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VALUATION AND GREENING THE NATIONAL ACCOUNTS: SIGNIFICANCE, CHALLENGES AND INITIAL PRACTICAL STEPS

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Valuation and Greening the National Accounts: Significance, Challenges and Initial Practical Steps[1]

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Abstract

Given that growing emphasis on non-market or environmental valuation in policy thinking, there is a real need to consider these developments in on-going efforts to green the national accounts. This paper reviews a range of issues that are implicit in taking up this challenge as well as tracing the beginnings of some of the steps towards practical valuation in the accounting domain. We begin by setting the scene in describing some aspects of the broader policy context that arguably make non-market or environmental valuation a relevant topic for consideration in the context of future directions in green national accounting. The discussion then proceeds to review in more detail some basic valuation concepts and outlines whether the extent to which non-valuation is consistent with national accounting practice. This includes a (very) brief overview of non-market valuation methods as well as a digression on benefit/ damage based approaches and cost-based approaches. Progress in valuation in the domains of health and ecosystems is also reviewed before taking stock of methods both on their own merits and in the light of the needs of accounting practice. There follows a consideration of a number of elements of how non-market valuation methods might be moved on to inform the needs of the national accounts. This discussion considers the use of value (benefits) transfers approaches as well as meta-studies to make sense of the diverse data that currently comprise the empirical record on non-market values and environmental values more specifically. Other aspects of the need for 'scale-up' are also discussed but it is argued remain (currently) at the research frontier. The final substantive discussion in this paper makes a tentative assessment of the scope for incorporating a range of different categories of environmental damage or service in the accounts including a discussion of whether data needs are currently met or where these remain incomplete (and to what extent).

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1. Introduction

The development of methods to value environmental (and related non-market) goods and services continues to evolve apace. So too, in a great many countries, has the practical uptake of these methods accelerated in both the framing and the design of environmental programmes and policies as well as other public actions that have environmental consequences. One policy-related domain, however, where this uptake has been largely conspicuous by its absence is national accounting. Thus, even experimental adjuncts to the core accounts have been relatively silent on practical issues relating to accounting for environmental values. In this paper, we hope to make the case that this, on one hand, is a missed opportunity but that, on the other, it is not too late to address this situation while still acknowledging that undoubtedly there are challenges ahead in advancing along this path.

Our starting point is the claim that official guidance on the greening of the national accounts appears to be running behind policy thinking and evaluation elsewhere. The reason for this span a range of concerns covering issues about whether we *should* value the environment, whether we *could* value it given the adequacy of methods currently at our disposal and whether, once over these two foregoing hurdles, the resulting values are consistent with established national accounting practice. An alternative standpoint, however, is that the sheer advance in the development of these methods combined with their increasing use in shaping policy decisions renders the first two of these arguments less pertinent to current debates about greening the national accounts (albeit with some notable exceptions). The focal question then becomes: what is distinctive or special about the national accounts such that the role of environmental or non-market valuation thus far has been downgraded and, more positively, to what extent do these concerns still hold? On the face of it, this seems a rather mild shift forward in focus. Nonetheless, it is a shift that hopefully would move discussion on in a purposeful way to more meaningful issues, notably how methods which have typically have been designed to inform policies or projects which involve small changes can be 'scaled-up' in a defensible way for the accounts.

The remainder of this paper is organised as follows. Section 2 seeks to set the scene by describing some aspects of the broader policy context that arguably make non-market or environmental valuation a relevant topic for discussion in the context of future directions in green national accounting. Section 3 then reviews in more detail some basic valuation concepts and discusses whether the extent to which non-valuation is consistent with national accounting practice. Section 4 provides a (very) brief overview of non-market valuation methods and includes a digression on benefit/ damage based approaches and cost-based approaches. Progress in valuation in the domains of health and ecosystems is also reviewed before taking stock of methods both on their own merits and in the light of the needs of accounting practice. Section 5 considers some elements of how non-market valuation methods might be moved on to inform the needs of the national accounts. This discussion considers the use of value (benefits) transfers approaches as well as meta-studies to make sense of the diverse data that currently comprise the empirical record on non-market values and environmental values more specifically. Other aspects of the need for 'scale-up' are also discussed but it is argued remain (currently) at the research frontier. Given these preceding discussion, Section 6 provides a preliminary assessment of the extent to which relevant and arguably empirically important categories of environmental damage or ecosystem service are 'ready' for valuation in (experimental) accounts. Section 7 offers some concluding remarks.

2. Valuation Now Matters: The Policy Context

A prominent contemporary theme in the appraisal of public policies is the quantifying in monetary terms of the intangible impacts of policy actions. For example, within the domains of environmental or health policy, it is increasingly recognised that these intangible impacts are likely to comprise a substantial component of the total benefits of policy interventions. However, many of these impacts are non-market goods or services (or bads). This means that the value that the public places on these impacts cannot be observed simply with reference to market information such as price and consumption levels. This has given rise to a proliferation of methods that have sought to uncover, in a variety of ways, the value of these 'unpriced' goods and services. Some of the more prominent of these methods have been around for a number of years but, in a growing range of countries, use has been increasing recently most notably in environmental policy.

Tellingly, Hecht (2005) claims that the arguments about the merits of and progress in non-market valuation have been heard and "have not convinced the accountants yet" (p203). The reasons cited for this in the literature on greening the national accounts are manifold. Concerns have been expressed about the ethics of environmental valuation (e.g. Peskin and Lutz, 1993; de Haan and van de Ven, 2007), the consistency of non-market valuation and valuation using market prices and the focus of the national accounts on economic activity rather than wellbeing (e.g. Harrison, 1993), discomfort with the 'subjective' judgements that non-market valuation entails (de Haan and van de Ven, 2007) as well as scepticism that valuation is 'too difficult' and about how robust non-market values are (relative to their market-based counterparts) (Stiglitz *et al*. 2009). In combination, this gives a flavour of the breadth of the critical debate. Not all of these positions are logically consistent with one another (nor, in fairness to those authors who have proposed them, are they claimed to be). Clearly, for example, a determination that valuing the environment is morally dubious involves a strong subjective judgement.[2]

These debates have obvious importance. Yet whether monetary approaches are worthwhile obviously remains an important item for meaningful dialogue between both statistical offices *and* policy departments. And, in this context, it is important to note a growing number of the latter are using environmental valuation ever more routinely. Moreover, this work often uses 'official' estimates of shadow prices, as an input to appraisal of projects, policies and programmes. This does not mean that this use of valuation, where it is present, naturally should be extended to green accounts but it surely establishes a basis for discussion given its apparent relevance in policy related contexts that green accounts hopes to cast light upon.

[2] Thus, Peskin and Lutz (1993) assert that: "Many services of the environment are too socially important to be determined by willingness to pay" (p152). But while this could well be an important argument for social decision-making, it is less clear what its implications are for accounting. Presumably, for example, there are market goods currently measured in the national accounts for which justice arguments could be found for a more equitable distribution. But this does not in itself downgrade the worth of information about the value of transactions in such goods.

A glance then at the extent to which policy-makers have begun to rely on thinking explicitly about the monetary value of environmental benefits (and costs) appears to indicate that, for many governments, the 'political test' (of the associated methods that facilitate this) has been passed (see, for a review, Bureau and Glachant, 2006; Dale *et al*. 2009). Put another way, valuation methods have been judged by many countries to be fit for the (policy) purpose that they largely have been designed for: i.e. social appraisal of the relative merits of actions that result in environmental improvements (or deterioration). Of course, not surprisingly, the indications are that uptake of valuation methods varies substantially across countries. Bureau and Glachant (2006) note that, in the context of energy policy, the official use of valuation ranges from relatively extensive in the USA and (to a lesser degree) the UK to relatively little, if at all, in France (although, in the latter, valuation is used more extensively to appraise transport infrastructure investment as well as more modest use in water and forest management). Dale *et al*. (2009) report a similar pattern for a wider variety of policy sectors and countries.[3]

National policy styles are clearly important for understanding these differences although according to Nielson (2009), within Europe at least, this is becoming less and less critical with the growing influence of the European Commission on the way in which regulatory appraisal is conducted across member states. In this context, an increasing emphasis on impact assessment has led to an upward trend (albeit from a low base) in putting into practice the principle of requiring that the impacts of Europe-wide directives and regulations be evaluated in some way. Interestingly, this has led a handful of emerging applications seeking to appraise large-scale policy actions such as, for example, the Water Framework Directive (WFD). The WFD requires Europes' waterways and waterbodies to have reached 'good ecological status' at least by 2015 and has led in a number of instances to significant efforts by EU member states' governments to understand the monetary value of the benefits of this regulatory initiative. A study by Baker *et al*. (2009), for example, summarises the extensive non-market valuation that has been undertaken, on behalf on Defra[4], to inform appraisal of actions to reverse water pollution in England and Wales. Both Dale (2009) and Delbeke *et al*. (2010) describe the European Commission's Clean Air for Europe (CAFÉ) programme as an area where non-market health valuation (of mortality and morbidity) has had a growing influence in developing an EU-wide strategy and associated targets (e.g. for reducing air quality related health impacts) and emissions limits across member states.

This use of non-market valuation is increasingly codified in official guidelines (see, for example, Government of Canada, 1995; US EPA, 2000; HM Treasury, 2003; European Commission, 2008). The UK case is particularly interesting as part of the regulatory impact assessment that must accompany policy proposals is a statement, to be signed by the relevant HM Government Minister, attesting to the veracity of the appraisal and, furthermore, that with respect to the policy outcome that: "… I am satisfied … that the benefits justify the costs". Needless to say, the existence of

[3] Silva and Pagiola (2003) takes a detailed look at how environmental valuation methods have been used to appraise more than 100 projects within the World Bank's environmental portfolio. This indicates is, there has been a significant increase in the use of environmental valuation in recent years to roughly one-third of these projects.
[4] The Department for Environment, Food and Rural Affairs.

guidelines or such statements does not ensure always that valuation is always undertaken (e.g. Hahn and Dudley, 2007). Nor do we suggest that the use of valuation always can be taken at face value. Indeed, policy-makers might use valuation for a number of distinct reasons (Dale *et al*. 2009). Some uses are confirmatory (in the sense of conforming to the 'textbook' view of how this information might be used): namely, informing the specific design or selection of policies or, more generally, in framing thinking about policy options. Other uses might be bound up with the political economy of policy-making where information on non-market values might be used either to justify decisions already made or conducted as a symbolic gesture only. Whether these latter two uses of policy evidence are unique to valuation is, of course, highly doubtful.

These reservations aside, official guidance across countries is increasingly establishing non-market valuation as a focal tool of evidence-based policy. In practice too the increasing prevalence of environmental economics training amongst those in government and related departments has led to such official guiding principles to be more than just empty rhetoric and wishful economic thinking (Bureau and Glachant, 2006). This is significant, in particular, as environmental valuation requires an input of time and effort in order to understand the underlying rationale, the technical details and just as importantly 'the state-of-the-art' in current thinking on these techniques (Pearce *et al*. 2006).

It seems reasonable to assert then that, given the balance of these policy developments, there is an initial case for thinking once more about how environmental valuation might inform subsequent rounds of discussion on options to green the national accounts. If not then, at least for those countries that are using valuation prominently, environmental extensions to national accounts will lack relevance to policy thinking. Of course, it might be argued – by this same logic – that downplaying this role might not matter for those countries where policy-makers appear to place little emphasis on explicitly knowing the monetary benefits of environmental policies. Yet further reflection might suggest that this is not altogether justified unless, that is, it is the case that evolving thinking in national accounting practice should only be reactive to current policy demands. This process also needs to anticipate trends and the informational demands of the future. One (not unreasonable) interpretation of the way in which resource and environmental accounting practice progressed in its formative years (some two decades or so ago) can be put in these terms. Thus, it was arguably the case that the demand for these green accounts did not come primarily from policy departments. Rather, the accounts that were developed in a number of countries in the early 1990s 'ran ahead' of policy demands (Hamilton *et al*. 1994) presumably, in large part, because of the future information needs were being anticipated in precisely this way.

At the same time, it is worth stating plainly that, logically, the likely best place for thinking about the role of environmental valuation in greening the national accounts, remains – for the foreseeable future – within *experimental* accounts outside of the core national accounting framework. There are a number of reasons for this emphasis. The national accounts, in many countries, have evolved over a period of decades. Comparatively, environmental (or non-market) valuation is a rather young discipline and serious and systematic investigation can be traced back less far. It stands to reason, therefore, that the core accounts have developed to a considerably more mature state over that longer period. Similarly, it is also to be expected that difficulties will be more evident in the younger (valuation) field than in its older and more experienced counterpart. Indeed, these 'difficulties' frequently are cited as obstacles to placing money values on a fuller range of policy impacts (see, for example, OECD, 2004). In the environmental context, this is often taken as saying that monetary values reflecting environmental impacts are just not sufficiently 'accurate' to be policy useful.

While 'difficulties' cannot be dismissed out of hand, whether this position *always* reflects genuine complexities or provides a 'flag of convenience' is another matter. In such respects, what crucially needs to be spelled out in clear and explicit terms is exactly what the hurdle of accuracy is.[5] That is, how accurate do data need to be in order to sensibly inform policy decisions or policy thinking (and, by this token, has all evidence and arguments that are used to inform policy been subject to the same yardstick). It also needs to recognised that ignoring environmental valuation (or a particular non-market method) obviously contributes next to nothing in terms of overcoming the apparent difficulties associated with it. The decisive issue is whether (current or future) policy needs indicate that these difficulties are worth confronting (rather than evaded). Yet, at the some time, circumscribing this activity in terms of a more experimental approach has a clear advantage of avoiding the undesirable extremes of either 'ruling in' or 'ruling out' altogether environmental valuation in greening of national accounts. This also allows scope for proper and critical reflection on genuine issues of substance that confronts the use of non-market valuation in this way.

3. Accounting for Non-market and Environmental Values

3.1 Basic Concepts

The current status of 'valuing the environment' in official green national accounting stands in obvious contrast to valuing natural resources and changes in those particular assets. The basic reason for this is simple enough. Natural resources, such as sub-soil assets, are typically traded in well-established markets and thus have observable prices that serve as an obvious guide to the value of depletion. Yet, as is well-known, correctly valuing the user costs arising from the depletion of a natural resource entails all sorts of (implicit) assumptions to be made (about future paths of prices and extraction rates) which may or may not turn out to be 'reliable' or 'reasonable' (see, for example, Atkinson and Hamilton, 2007). However, the fact remains that many seem content with choosing between assumptions and making judgements in this particular context.

The same contentment cannot be claimed, within for example the national accounting literature, for the mix of assumptions and judgements that must be relied upon if a broader variety of natural assets are to be valued in the accounts. To an extent this is understandable. It creates a very real dilemma as Stiglitz *et al*. (2009) recently have identified when they state that: "In standard national accounting practice, the

[5] As an example, Nielson (2009) points to the dominance of producer interests in particular policy sectors – notably agriculture – such that there is an incentive among some stakeholders to oppose such approaches given that in a very real sense these values might be the liabilities of tomorrow.

normative issue of defining preferences is generally avoided through the assumption that observed prices reveal the true preferences of people. No explicit normative choice is therefore to be made by the statistician. But as soon as we recognise that market prices cannot be trusted, alternative imputed prices must be computed, whose values will strongly depend upon normative choices." (p75). And while 'choices' are not absent in the commercial natural resources case, the reasons for divergences between the empirical findings of the different methods that exist to measuring e.g. user cost (rather than the deeper issue concerning which method is 'correct') are relatively well understood at least. By contrast, what environmental valuation typically refers to is the collective efforts represented by approaches which seek to assign shadow prices in contexts where (direct) markets do not exist at all. What is of interest then, in these approaches, are *non-market* goods or services. These, in turn, might be public goods (or goods or services which generate externalities for third parties) and so provide benefits more widely across individuals or households whether or not they have bought by these economic entities (Nordhaus, 2006).

There are a number of different elements – and so too implications for the national accounts – to what non-market aspect of the economy needs to be valued here.

First, the environment (and so natural assets) clearly supports economic activity in a number of important ways. In this sense, the environment is a non-market input to production and consumption that is captured in the national accounts as they currently stand. The issue is that this contribution is not attributed to its correct source (Nordhaus, 2006). So, for example, environmental damage might result in lower production – perhaps because of the exposure of crops to ground-level ozone – and possibly also lower capitalised land values although these losses do not explicitly enter the accounts. In such cases, the environment is a non-marketed input but its value can be understood in terms of goods and services which do command market prices.[6]

Secondly, economic activity generates environmental damage which results in impacts which (currently) lie beyond the market. These impacts are numerous although significantly in the case of air pollution will comprise health (both in the form of premature death and increased incidences of e.g. respiratory illnesses) amongst other effects. Pollution of water bodies such as rivers and lakes similarly will have a range of impacts on visual amenity, aquatic life and (again) human health for those exposed to polluted water. In all of these cases, what is lost when the pollution impact occurs might hardly be felt (if at all) in the form of any market transaction (although this is, as ever, always an issue of extent).

Thirdly, for examples such as water pollution, the other side of the coin is that what is being enjoyed are outputs in the form of goods and services which are provided by natural assets such as, in this instance, an ecosystem. In large part, these goods and services might be non-market in nature. Just as importantly, pollution might not just disrupt the enjoyment of a non-market service but might damage the underlying asset and thus the ability to provide future services (at some level) as well. Clearly, ecosystem destruction through e.g. permanent land conversion will have much the same effect as well.

[6] Although, in turn, these observed prices might be distorted by the impacts of subsidies and so on.

In any of these cases, however, policy might eventually act so as to establish an explicit price through pollution taxes or those instruments such as 'payments for environment services' or emissions permits which can be traded freely (see, for example, Engel *et al*. 2008; Hanemann, 2010). In the case of the latter, for example, where these markets are mature and extensive the prices that we observe might be used as a reliable guide to the value of a good or service provided by the environment or the damage caused by a unit of pollution being released in some receiving environment (Dietz and Fankhauser, 2010). These emerging markets (while mostly in their infancy) are clearly of relevance to our current discussion. However, for the most part, our interest here – in the interim – is framed by the role that accounting for non-market values associated with the environment can play in greening the national accounts.

On the face of it, this emphasis on values or prices which lie beyond the market is very different to any framework premised on market prices. Clearly, for example, measurement raises distinct challenges. Yet it is important also to recognise that at some basic level, the concepts employed are essentially comparable to the case where a market price exists for the good or service in question. That is, when – in environmental economics – a non-market value is sought for an environmental impact, what is being measured is not something that is completely at odds with the market price approach or, for that matter, national accounting practices. Needless to say, however, there are nuances to this assertion.

To begin with, what is potentially to be valued is a far wider range of benefits (and associated sources of wellbeing) than say for commercial natural resources. This should not be a surprise as what is at stake is a far broader range of natural assets (and thus goods and services that are provided by these assets and, so too, the reasons why these might be valued by people). Within environmental economics, the concept of total economic value (TEV) is used a convenient organising framework for thinking about these different sources of value. Moreover, it is usual to divide this notion of TEV into use and non-use (or passive use) values. Use values relate to actual use of the good in question (e.g. a visit to a national park), planned use (a visit planned in the future) or possible use. Actual and planned uses are fairly obvious concepts, but 'possible use' could also be important since people may be want to maintain a good in existence in order to keep open the option of using it in the future. Non-use value refers to the motive to maintain some good in existence even though there is no actual, planned or possible use (e.g. to preserve an ecosystem as a bequest for future generations or simply for its own sake). Within the economic framework, some of these values plausibly may spill over to the residents of countries additionally to those who live within the same national boundary as the natural asset in question.

In all of these cases, what is to be valued is the willingness to pay (WTP) of individuals for a marginal change (e.g. an increase) in the provision of an environmental good or service on the basis of use, possible future use or non-use.[7]

[7] This is not the only way of looking at the way in which an individual might value a change in provision. That is, for an environmental improvement, the change in wellbeing that an individual enjoys can be measured by his or her WTP for, or his or her willingness to accept (WTA) compensation to forego, that improvement. For a corresponding (but opposite in sign) environmental deterioration, the change in wellbeing is the WTP to avoid that outcome or WTA compensation to

Crucially, of course, this WTP is not the same as the price that has to be paid and the literature on e.g. environmental valuation a firm distinction is drawn between 'value' and 'price'. Bateman *et al*. (2010) illustrate this using the example of water which they describe as: "… the stuff of life without which existence is impossible. Yet the price we pay for water in our household bills is actually very modest. It is clear to see that 'value' and 'price' are not necessarily the same thing. In fact, price is simply that portion of underlying value which is realised in the market place. Now in many cases price may be a perfectly acceptable approximation to value […] However, as the water example shows, market price can be […] a poor approximation to value, indeed this divergence can often be substantial and is a characteristic of many of the goods produced by the natural environment." (p16).

Essentially, there are two related dimensions present in this description. On the one hand, 'value' refers to social value: i.e. a market price only captures part of is socially important as a result of the provision of a good or service. On the other hand, there is an issue about what the total value of provision is: i.e. even if the market price was a good approximation of social value, what economists typically are interested in lies also in knowing something about the consumer surplus (the enjoyment derived from consuming a good over and above the price paid for it). Both of these dimensions of the distinction between price and value raise issues about the extent to which non-market valuation is consistent with long-established principles of treatment of valuation within the national accounts. It is to this issue to which we now turn.

3.2 Non-market Values and the National Accounts: Consistency or Mismatch?

Another long-standing impasse in discussions about environmental valuation and the accounts is the alleged inconsistency between the conventional accounts, primarily based as they are on market prices, and approaches which generate values which lie outside the market (or perhaps are only indirectly reflected in market transactions). Harrison (1993), for example, expresses these concerns in the following: "Adjustments to be made to the SNA are […] a move toward a measure of welfare … This is not mixing like with like, however. If a true welfare measure were to be derived, then the basis for valuing all the other transactions recorded in the SNA at present would need to be examined and in many cases would need to be altered" (p72) . There are probably at least two elements to this critique.

The first element of concern arguably boils down to fundamental disagreements about the 'spirit' of the (existing and established) national accounts – i.e. to measure economic activity – and the 'spirit' of valuing the environment and (non-market aspects of) environmental change – i.e. to measure (human) welfare or wellbeing. It follows therefore that bringing these two perspectives together is one possible example of 'not mixing like with like'. Of course, this is only one way of looking at the information in the national accounts (and begs the question of what we are measuring economic activity *for*). An alternative, following for example Nordhaus (2006), is to say that notions of national output and income provide the bases for saying something about economic welfare. The issue is that it is these magnitudes are

tolerate it. Choosing between WTP and WTA, in a given context, remains a contentious issue although in large part the choice is determined by whether people affected by a policy have a property right to the current or the new level of provision of the good in question.

an incomplete description (and perhaps grossly so) of this object of interest. Bringing the value of non-market impacts within the ambit of national accounts is thus one means of working towards a more inclusive notion of economic welfare. This can be seen as the practical counterpart of contributions to the economic theory of green national accounts (e.g. Hamilton and Clemens, 1999; Dasgupta and Maler, 2000; Hamilton and Withagen, 2007; Dasgupta, 2009). In all of these contributions, resolution of the asset accounting problem serves the end of providing a better guide for the social welfare ('embodied' in current wealth) and wellbeing at different points in time.

This emphasises that there are different traditions upon which green national accounting is premised and, as a result, different audiences for the empirical outputs of this practical work even if an alternative tradition is an anathema in some quarters. Moreover, an immutable starting point that the national accounts are only about economic activity, by definition, is hard to reconcile with the way in which environmental policy thinking beyond the accounts has evolved. Put another way, concern for the consequences on broader wellbeing of more narrowly construed economic activity is where the motivation for understanding more about non-market impacts begins. The fact remains that just because a non-market asset such as an ecosystem does not command a price, this does not mean that its services have no value. Indeed, it is this absence of an explicit value that increases the prospects that too many ecosystems will be destroyed or damaged. Non-market valuation is thus the bedrock in the provision of information that can be used to frame and design policies that, in due course, might correct such market failure.

On a strict view then of what the national accounts stand for, environmental policies seek to improve something (i.e. broader wellbeing) which, ultimately, must remain outside of these accounts. This, in turn, restricts measurement and analysis to looking only at the burden that economic activity places on physical environmental endpoints or, in turn, the burden that environmental policies place on measured economic activity. In either case, this only goes part of the way to engaging with what policy-makers are actually interested in. A critical issue is whether this apparent dividing line between what is economic activity (the domain of the accounts) and what is not is worth preserving *whatever the cost* in terms of building a bridge to contemporaneous thinking about environmental policy. In other words, it might be valuable to be say something about this (environmental) link between economic activity and broader wellbeing even if we wanted to preserve the strict emphasis of the core accounts on the former.

A second prominent aspect of 'not mixing like with like' is the claim that non-market values are not like market prices which lie at the heart of valuation in the core accounts.[8] The specific issue is said to be that these non-market values measure consumer surplus and as such, it is argued, represent a basic and unacceptable inconsistency. Nordhaus (2006) puts it thus: "… to introduce consumer surplus in the augmented accounts would introduce a major inconsistency … because the standard accounts are based on marginal values" (p184). Yet while there is a genuine basis for this concern, its general implications arguably are overstated.

[8] Clearly, this is a little misleading. In almost all countries, a significant part of what is measured in the national accounts is government activity where output is not valued using market prices and, indeed, 'output' is not really measured at all.

Specifically, what this concern reflects is the feeling that what is to be added into national accounts is a measure of the total value of a non-market good and service. This, in effect, corresponds to the value that is estimated by measuring the entire area under a demand curve (for that good or service) between some reference point – such as the 'zero' level of provision – and the current level of provision (Nordhaus, 2006).[9] This total value thus includes the consumer surplus that people enjoy from consuming all units of that good or service. Typically, cost-benefit appraisal – perhaps the primary use for non-market valuation – would be interested in small changes in the provision of a service (or a change in the price of the service). What is being measured as part of the benefit of that change is the change in consumer surplus between the existing and new provision of the service (or the current and new price). This is not an issue about non-market valuation as such. Consumer surplus (or, more specifically changes in it) is also the locus of a (cost-benefit) appraisal of policy actions that increase the provision (or change the price) of a market good.

The crucial distinction to be made here – to reconcile the 'economics' and the 'accounting' –is between total value and marginal value (Nordhaus, 2006; Abraham and Mackie, 2006). The latter is what people would have been willing to pay for the last unit of the good or service that is consumed. It is this price which – for national accounting purposes – should be multiplied by the quantity (or quality) of the good or service consumed. Thus this issue boils down to whether non-valuation techniques are able to estimate something akin to a 'conventional' demand curve for a market good and, in doing so, describe willingness to pay at different levels of 'provision'. At least two comments can be made in respect of this:

First, in principle, there is no inconsistency with valuation using market prices in that all of the major techniques of non-market valuation can be shown in theory to estimate marginal values (see, for example, Freeman, 2003). Hence, these approaches can be used to say something about the price that people, in principle, would pay at the current level of provision of a non-market good or service. Clearly, this is not exactly the same thing as the price that would prevail were a market established. But nonetheless, non-market valuation methods provide information which offers a guide to marginal WTP in an analogous way.

Secondly, examples can be found where practical valuation studies have sought to estimate a schedule of marginal WTP values that would suit the informational needs of the accounts. However, an interim comment is that it remains worthwhile exploring further how many studies in the literature in practice present data on non-market values in a way that would be immediately usable in national accounts. That is, reported findings more often than not could be geared towards the informational needs of economic appraisals. And, moreover, reported practical WTP estimates might be interpreted as marginal values but only under certain assumptions.

We comment in a little more detail about some of these issues with reference to three specific valuation methods. A study by Mourato *et al*. (2004) of the willingness of pay of visitors to the Machu Picchu Historic Sanctuary in Peru provides an illustration of

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[9] In fact, the reference point does not need to be zero. It could be some other level such as the level of carbon sequestration services that would be provided, for example, if land currently under standing forest – hypothetically but credibly – was pasture instead.

using non-market valuation to infer something about the demand curve for a good. In this particular example, however, the good itself is not strictly speaking non-market in that an entry price was charged to visitors (and so presumably recorded in the national accounts in some way). However, for a variety of reasons, these institutionally set prices provided little guidance to what visitors would be willing to pay for the marginal trip. This study sought to use elicited valuation data on the distribution of WTP in order to construct a ‘demand curve’ for visits. This indicates the marginal value that a representative visitor places on a trip at a given level of visitation. These findings were also used to simulate what is the marginal WTP or entry price that would prevail under pricing regimes which mimicked market-like criteria (such as profit maximisation).

Siikamaki and Layton (2005) provide a comparable exercise in addressing the question of what is the optimal level of conservation in Finland. The demand side provided estimates of the marginal value (i.e. the marginal WTP) of Finnish citizens to conserve forestland at different levels of provision where ‘benefits’ were largely non-market. The supply side, or the marginal costs of conservation, is measured through the estimation of incentive payments that would be sufficient to compensate land users for the opportunity costs of conservation.

In both instances, these studies were interested in a rather different question to one that might be posed in the context of greening the national accounts. Specifically, in Siikamaki and Layton (2005), they ask how much forestland – currently part of a conservation network – would it socially desirable to protect and, in Mourato *et al*. (2004), what prices would visitors face if heritage managers were permitted to capture a greater proportion of the benefits that tourists enjoy when visiting Machu Picchu. One study which claims to talk directly to the issue of the consistency of non-market valuation studies and the accounts is that of Campos and Caparrós (2007). In their case, they look at recreational demand in the (unpriced) woodlands of Extremadura in Spain. Nevertheless, the procedure is in many the same as in the two previously discussed studies in that it uses simulation procedures to generate market-like prices from non-market data about, in this case, forest recreational demand.

What these studies do is to estimate what the likely level of provision and, in doing so, the likely prevailing price or marginal value that would arise if that service or those services were provided at some level by the market. This is interesting but clearly brings another hypothetical dimension into the reckoning: the simulation of what price is likely to prevail if something that resembling a proper market was established. However, the basic point stands about the broad consistency of non-market valuation methods with existing valuation method used to compile the national accounts.

Ultimately, emerging markets in payments for ecosystem or environmental services might be used be used to establish the prices to attach to relevant changes in quantity or quality. Similarly, markets in carbon (and other polluting substances or activities subject to overall limits and trading schemes) might serve as such as guide. It would be useful to collate the empirical record on the values that currently emerge in all of these trades and markets. However, we are probably still a little way off (with some exceptions) being able to use these values for accounting (and related) purposes (see, for example, Engel *et al*. 2008). Interestingly, for example, it appears that no meta-study of these transactions is as yet available (although a variety of data are available through currently disparate sources).

In principle at least it would seem that the basic issue of consistency – that values in the national accounts do not include consumer surplus and nor must non-market values if these similarly are to be included – is not violated as long as valuation is based on measuring provision at the marginal value that people place on it (Nordhaus, 2006). Thus while this particular concern about 'like with like' remains a useful point in guiding how non-market values are to be presented in the context of greening the national accounts, contrary to assertions which arise periodically, it does not justify the application of a brake on the entire enterprise.[10] Specifically, what we must seek to measure for stocks and flows (including changes in stocks) on this basis is – in broad outline – the following.

- *Current services*: for an environmental service (or good) consumed in some current year, what we need to measure is the product of the amount of the service is consumed and its marginal value (i.e. what people are willing to pay to consume the service at that level of provision).
- *Assets*: the value of the natural asset or stock that gives rise to this service is the present (i.e. capitalised) value of the services that it will provide again valued according the marginal value of these services.[11]
- *Asset change*: the change in the stock similarly can be measured as the change in the present value of the services that are provided as a result of some current increase or decline in that stock (e.g. shrinkage of an area of wetland).

The challenges here are in large part those relating to practicalities. These include (but are not restricted to) how to discount future services (in the case of asset accounting) and – beyond the discounting dilemma – determining what the value of those future services might be. Similarly, decisions about which aspects of total economic value are relevant to the national accounts as well as how to handle international spillovers (as in the case where people living far away from a resource such as a tropical forest derive some enjoyment from its conservation) are also important. Clearly, there are

[10] This emphasis on marginal values or WTP by putting non-market valuation on more of a par with valuation in the national accounts also avoids what Nordhaus (2006) terms, the 'zero problem'. This arises if the valuation includes all the entirety of the wellbeing that is enjoyed by every unit of the good or service that is consumed. Put simply, the question then becomes what is the total value of that service compared to its (complete) absence? If the good is essential (that is, some basic level of provision must be available for e.g. survival then the total value will tend to infinity). Concern about this issue was evident in much of the debate about that followed Costanza *et al*. (1997) on the total value of global ecosystem services (see, for example, Bockstael *et al*.2000). This issue has re-emerged once more in large scale (official) economic assessments of national and global ecosystems. Nordhaus (2006) notes that this is not just a problem for non-market valuation but rather it relates to any essential input whether bought and sold in markets of not (e.g. the total value of consuming all of the food that has a nation buys in a particular year as opposed to starving).

[11] Nordhaus (2006) argues that it may not be sensible to seek to measure the whole of the stock in this way. One element of this reasoning is that these values might otherwise overwhelm – especially if services are discounted over long perhaps infinite time horizons – (and be inconsistent with the treatment of) other existing assets in the accounts. As a result, he proposes measuring the stock relative to some base year. Ultimately, if this base period is last year's stock then this boils down to asking what the net accumulation or asset change over the accounting period. In all likelihood, however, it is this which is probably single most policy relevant information that non-market extensions to the national accounts can provide.

also concerns about the standing or validity and reliability of values that emerge from non-market valuation studies as well as the ability of these methods to handle complexities particularly those concerning the nature of ecosystems. The remaining sections of this paper will focus in more detail upon some of these areas of concern.

4. Issues in Non-market Valuation and the Accounts

4.1 A Brief Overview

As is well-known, there are a good handful of candidate methodologies that have emerged as ways to value non-market goods and services. The merits of each of these methods depend in large part on the nature of what it is that is to be valued and, in particular, the extent to which clues to value can be detected in market behaviour or not. So, for example, some of these approaches estimate original values perhaps by looking at *actual* behaviour – i.e. revealed preference (RP) – or *intended* behaviour – i.e. stated preference (SP). The former (RP) looks at 'surrogate markets' and, in doing so, infers preferences for non-market goods as implied by past behaviour in an associated market. In other words, these methods seek to quantify the market trace of non-market goods and services or, for example, how these benefits are affected by environmental degradation. The latter (SP) is an umbrella term under which are found a range of survey-based methods that use constructed or hypothetical markets to elicit preferences for specified policy changes. By far the most widely applied SP technique is the contingent valuation (CV) method. However, in recent years, choice modelling has become increasingly popular.

All of these approaches have been applied extensively over the past two decades or more to value an ever growing number and variety of natural assets. In the case of RP methods, applications here have utilized the fact that the (complementary) purchase of market goods is typically required to access a recreational area (e.g. parks, woodland, beaches, lakes etc.) and that this insight – through so-called travel cost (TC) approaches – can be used as a basis for valuation of these particular assets. By contrast, avertive behaviour and defensive expenditure approaches determine value by looking at the purchase of substitute goods that help avoid negative intangible impacts. Thus, people buy goods such as safety helmets to reduce accident risk, double-glazing to reduce traffic noise and, in doing so, reveal something about their valuation of these bads.

Other market behaviour based approaches take as their starting point the observation that market prices can be a function of a bundle of characteristics which might include dimensions such as environmental quality which lack an explicit price. The hedonic price (HP) method uses statistical or econometric techniques to isolate the implicit 'price' of each of these characteristics in two types of markets: (a) property markets and (b) labour markets. In the former, hedonic studies of the property market have been used to identify the value of non-market goods (or bads) affecting house prices such as road traffic and aircraft noise, air pollution, water quality, proximity to landfill sites and planning restrictions on open spaces in and around urban areas. The HP method has also been used to estimate the value of avoiding risk of death or injury by looking for price differentials between wages (employees' willingness to accept compensation) in jobs with different exposures to physical risk (Taylor, 2003; Krupnick, 2004)

Production function approaches look at the value of natural assets and environmental services from the perspective of their affect on the costs and output of producers: that is, valuing the environment as an input (Hanley and Barbier, 2009). So in the case of applications to agriculture, the production function approach begins with the assumption that farmers produce crops that, for example, are irrigated with groundwater from accessible aquifers. Thus the amount of a given crop produced is a function of water (which is itself a function of the groundwater level) along with other inputs. A decrease in groundwater levels would result in an economic loss determined by the either an increase in pumping costs or a change in productivity. Specifically, in the former case, as groundwater falls below a certain level and pumping slows down, the costs of pumping water are likely to rise in the form of extra hours needed to continue that activity. In the latter, if groundwater levels fall below the maximum depth of tubewells then the farmer ceases pumping and agricultural production will fall.

Applications of the production function approach to this type of problem include Acharya and Barbier (2000) for the Hadejia-Nguru wetlands in Nigeria. Other examples include soil erosion (Barbier, 1998). Of course, the distinguishing feature of this approach is that it is measuring something that is likely to be captured in the national accounts already. The non-market aspects of the method comes from the fact the environmental input (soil, groundwater and so on) may be unpriced and thus the factor is not paid for (Hanley and Barbier, 2009). Extensions, however, include the assessment of a wide range of values enhanced by the provision of storm protection services in the case of natural assets such as coastal mangroves where the benefits in question include a mix of market and non-market outputs (such as, in the case of the latter, health protection).

In SP studies, using a questionnaire, a hypothetical (or contingent) market is described where a non-market good or service can be traded. The method can be used, in principle, to estimate *all* the benefits – use and non-use – associated with a change in the level of provision of a good or service. Not surprisingly, therefore, the range of environmental issues addressed so far by this approach is wide: e.g. water quality, outdoor recreation, species preservation, forest protection, air quality, visibility, waste management, sanitation improvements, biodiversity, health impacts, natural resource damage and environmental risk reductions to name but a few. Carson's (forthcoming) bibliography of published and unpublished CV studies contains over 5,500 studies, undertaken in just under 100 countries. The application of choice modelling in environmental economics is a more recent development with the most widely used variant being the choice experiment (CE) technique (see, for example, Bateman *et al*. 2002, Hanley *et al*. 2001). In a CE survey, respondents are required to choose their most preferred out of a set of alternative policy options. For example, in the case of quality improvements in some water body such as a river, the attributes might be a boost to aspects of the river's ecology (perhaps as indicated by fewer fish deaths), decreased health risks to those who are exposed to the water such as swimmers and rowers and increased visual amenity. A price or cost variable is typically one of the attributes and non-market values can be indirectly inferred from the choices made.

In large part, the case for (and against) all of the methods have been well-rehearsed elsewhere in the environmental economics (and related) literature as well as policy documents (see, for example, Bateman *et al*. 2002, Champ *et al*. 2003, Habb and McConnell, 2002, Alberini and Kahn, 2006 and, in the context of ecosystem valuation, US EPA, 2009). Those reviews, and the issues that they identify, are relevant here but our primary focus is on those aspects, and surrounding debates, that are most relevant to greening the national accounts. A critical element of this discussion then would naturally be the extent to which these methods are actually measuring environmental dimensions which are implicitly captured in the national accounts already either in current measures of output or asset values. Clearly, this is not a concern for typical applications of these methods where the focus is the appraisal of proposals for environmental improvements (or simply to understand the value of environmental damage in some 'stand-alone' ecosystem assessment for example). From the standpoint of incorporating these values into national accounts, however, the prospect of double-counting clearly comes to the fore. For some of these valuation methods, the potential seems obvious: for example, and as previously mentioned, production function methods where non-market values are inferred from looking at a market good. But the issue is perhaps rather less the method than the nature of the good or service which is being valued. A thorough identification of this potential for double-counting remains an important practical issue for future work; that is, once it is determined that non-market valuation methods have a place in continuing efforts to the green the national accounts.

We take stock of the progress made in the field of non-market valuation and its implications for the accounting domain in what follows. Before proceeding to that point, however, a noteworthy element of all of the methods so far (briefly) discussed is that they seek to estimate the money value of the benefits of environmental improvements or, conversely, the value of the damage caused by environmental losses. Within the national accounting domain, a prominent argument has centred on whether this benefit (damage) focused approach can be replaced with an alternative approach on cost. The next section, therefore, considers these issues in greater detail.

4.2 Benefits/ Damage vs Restoration/ Maintenance Costs (Revisited)

Within the accounting setting, a choice is often presented between those methods which seek, for example, to value the *damage* that arises from the depletion or degradation of natural assets (or conversely the *benefits* of improving those assets) and those methods which seek to value the *costs of replacing* or *restoring* these same assets to some level. These competing standpoints have been evident in early debates about accounting for the value of air and water pollution as well as, more recently, in deliberations about ecosystem accounting.

Interestingly, this debate about the merits of one approach over the other has little parallel outside of the national accounting domain. In policy terms, both of these informational elements provide critical evidence about the costs and benefits of actions: i.e. respectively by asking how much it might cost to reverse environmental decline and whether that burden is worth incurring. But in terms, for example, of accounting for some decline that arose in the first instance what crucially we need to know is the value lost because of that damage. Barbier (2007), in the context of ecosystem valuation, puts the problem thus: "Herein lies the main problem with the replacement cost method: it is using ‘costs’ as a measure of economic ‘benefit’. In economic terms, the implication is that the ratio of costs to benefit of an ecological service is always equal to one.” (p194). Of course, some proponents of this approach might not see this necessarily as a problem. The suspicion must be that these commentators judge that the decision to conserve or restore is always the correct one and the emphasis on costs clearly serves that purpose.

Within the context of national accounting, the use of restoration or maintenance costs recently has been proposed in the context of ecosystem accounting in a number of related papers from the European Environment Agency (EEA). Specifically, it is proposed that the degradation of ecosystems can be valued in national accounts with reference to the costs that would need to be incurred if all of this loss was to be restored. A key element of the defence of this approach outlined by, for example, Weber (2009) draws a parallel with the traditional practice of accounting for the depreciation of produced capital. This ‘consumption of fixed capital’ as it is termed in the UN System of National Accounts is valued as the amount that it would cost to maintain this capital used in production, given estimated profile of its depreciation, based on the value of new assets. Depreciation, in this sense, then can thought of as the costs that would be incurred in keeping the stock intact. By analogy, Weber argues that is what good for the treatment of produced capital in the national accounts is also good for the treatment of the degradation (i.e. depreciation) of ecosystem assets.

It can be argued, however, the analogy does not work completely. Thus, United Nations (2008) in setting out how depreciation is treated in the current SNA states that: “Consumption of fixed capital is […] determined by […] the benefits that institutional units expect to derive in the future from using the asset in question over the remainder of its service life.” (p124). And the justification for the focus on replacement or maintenance cost is spelled out as being that: “Conceptually, market forces should ensure that the purchaser’s price of a new fixed asset is equivalent to the present value of the future benefits that can be derived from it.” (pp124-5). For an ecosystem asset whose services are predominately non-market in nature, of course, this relationship is unlikely to hold. In other words, the choice of a cost-based valuation method in the case of a non-market asset such as an ecosystem cannot be justified on the logic that this calculation is similar to the value that we are actually interested in (i.e. the present value of the services that are lost when ecosystems are degraded over the accounting period).

In the case of replacement costs (e.g. the costs of replacing a service by an alternative method such as a water treatment plant in the case of the water purification currently provided by an ecosystem), Barbier (2007) cites an earlier contribution by Shabman and Batie (1978) about when this method is reliable (as a means of valuing the benefits provided by e.g. this ecosystem service). These are three-fold and require that the alternative: (a) provides the same service; (b) is the least-cost option; and, (c) there are strong reasons to believe that society would demand the service that would be provided by this least-cost option. This represents a useful checklist or ‘burden of proof’ for those who advocate replacement costs method (or maintenance costs given that the implicit assumption there is that restoring the asset is the least cost option to ensuring a service is continued). One conclusion is likely however. This check is unlikely to favour the cost-based approach being reckoned to be suitable for application across the board to all ecosystem assets and, in all likelihood, only a relatively small sub-set of these assets.

Thus, the narrow set of circumstances in which the costs of restoring or replacing that a non-market good and or service coincide with its benefits effectively rules out a cost-based approach as a valid method *at least for this purpose*. This is unfortunate because the uncertainty around estimates of benefits or (conversely) damages is likely to be somewhat greater than those around costs. This is the tack taken by Dietz and Fankhauser (2010) in the context of target-setting (in emissions reductions strategies) in climate change policy. In doing so, they stress the relative uncertainties of estimates of the costs and the benefits (i.e. the foregone damages) of the reducing carbon emissions, at the margin. In particular, they argue quite reasonably that uncertainty around the former is likely to be considerably less than for the latter. Moreover, it is probable that this is the case in the context of ecosystems as well.

The authors' specific concern is which (if any) is the better basis for establishing the shadow price of carbon defined as the price that will deliver the target that has been set. In this respect, it is argued that the implication of uncertainty is that this value might be more sensibly benchmarked against the more certain of these two parameters. That is, if the overall goal for climate change has been established – such as the often cited stabilisation of atmospheric greenhouses gases at 450ppm – then, under this proposal, the shadow price of carbon is set at the marginal cost of reaching this target. Actions to reduce emissions of greenhouses gases then can be appraised in the light of whether they involve (marginal) costs above or below that value. This approach to the problem of setting an 'official' shadow price of carbon has gained some ground in, for example, the United Kingdom.

It is tempting to conclude then that this reduces considerably the role, if any, that assessing benefits (or foregone damages) might play in this analytical process. However, that conclusion would be erroneous as Dietz and Fankhauser (2010) are keen to point out. In fact, they argue there remains a critical role for this other side of the coin. That is, it is difficult to see how, in this case, climate targets can be set without reference to and information about the benefits of emissions reductions. Rather it is the case, and reiterating an earlier point, that both sets of information are needed to frame policy choices sensibly even if one element of that choice is more uncertain than the other. In a very real sense, what the cost-based approach – and indeed any approach eschews any proper benefit or damage valuation does – is ignore something that is important to decisions. In other words, disregarding this information (perhaps because it is too 'uncertain') does not necessarily increase the certainty that a good decision is made if what is ignored is a critical factor that should be brought to bear on how to make decisions (Pearce *et al*. 2006).

The issues are much the same in the accounting setting given that this information ultimately is sought to improve future decisions based on a better understanding of what is lost from, for example, accounting for ecosystem damage. To be fair to the EEA position it appears to be only in the case of their proposals on ecosystem accounting that the emphasis is solely on replacement cost. Elsewhere there are clearer indications, particularly in the research on wetland valuation undertaken for the EEA, that a rather broader basis for valuation including benefit estimates (derived from revealed and stated preference studies among others) is being explored. We discuss this work in more detail in Section 5.2.

As a practical matter, the evidence is that any exclusive focus on replacement costs is likely to result in perverse policy decisions. As an illustration, Barbier (2007) provides a comparison of the value of storm protection services in Thailand using the replacement cost method and a damage valuation method (based, in turn, on an extension to the production function approach which sought to estimate how changes in coastal wetland area lead to changes in the incidence of natural disasters that cause economic damage).

Table 1: The Value of Storm Protection in Thailand, 1996-2004, (US$, millions)

	Average annual mangrove loss	
	FAO estimates	Thailand estimates
Replacement cost method		
Annual loss	$25.5m	$4.9m
Net present value (10%)	$146.9m	$28.0m
Damage cost method		
Annual loss	$3.4m ($2.3m – $5.8m)	$0.6m ($0.4m – $1.1m)
Net present value (10%)	$19.5m ($13.5m – $33.4m)	$3.7m ($2.6m – $6.4m)

Source: adapted from Barbier (2007)

Table 1 outlines the main findings of this comparison. The results are interesting for at least two reasons. First, there are two estimates of the physical extent of the mangrove loss along Thailand's coast. Thus, estimates published by the UN Food and Agricultural Organisation (FAO) indicate (annual average) rates of loss which are over five times greater than estimates published by the Royal Thailand Forestry Department (respectively $18km^2$ and $3.44km^2$). This indicates that uncertainty is important but is not just confined to valuation.
Secondly, in this case study, the replacement cost approach is significantly higher than the damage method. The latter values are specified as a range although in all cases the replacement cost values fall well outside of this array.

4.3. Valuation in Practice: Empirical Applications

As the development and use of valuation methodologies has evolved a vast amount of information has been generated about the value that people place on a wide range of environmental changes. Again, this emerging empirical record in all of these domains has been reviewed extensively elsewhere. However, in order to provide some flavour of the state of the existing evidence base, we review (relatively briefly given the breath and depth of the associated literatures) two prominent areas of valuation activity: human health and ecosystems.

The reason for this dual focus is that arguably the most experience and understanding has been garnered in respect of the former. Thus, relatively speaking, health valuation is a mature area of the discipline. To the extent that we know about where valuation works (and where problems exist) it is in this context of understanding the value that people place on reducing mortality risks and morbidity. This progress is frequently reflected in the adoption by numerous countries of single (or ranges of) values for their official mortality (and morbidity) values which are used in policy appraisal. Just as importantly, repeated studies indicate that the values that the public places on health improvements is comparatively large and, for example, assessments of the social costs of air pollution, for example, typically are dominated by this category of damage. By contrast, progress in ecosystem valuation has been more recent although significant efforts have been directed to this topic in recent years. The indications are that developments in the empirical record have been very real but, at the same time, so have apparent methodological challenges.

Progress in Valuing Human Health

Environmental policy affects human health in a number of ways. First, by reducing environmental risks to lives, it may 'save lives', i.e. reduce premature mortality. Second, it may improve the health of those living with a disease such as a respiratory illness and so result in a morbidity benefit. Valuing these mortality and morbidity effects in monetary terms can provide extremely useful information for policy. Past evidence indicates that the benefits of reducing human health effects (mortality and/or morbidity) often exceed the costs of pollution control by considerable margins. Over the last 30 years, stated preference studies, together with revealed preference methods, have been used extensively to calculate both individual WTP to secure reductions in the risk of death arising from a policy and WTP to avoid particular health outcomes. In this time, substantial progress has been made particularly in the valuation of mortality risks.

For convenience, WTP for mortality risk reductions is normally expressed in terms of the value of statistical life (VOSL). This implies dividing the WTP for a given risk reduction by that risk reduction to obtain the VOSL (see, for example, Bolt *et al*. 2005; Krupnick, 2004). Various countries adopt single (or ranges of) values for the VOSL and then use them in policy appraisal. The US Environmental Protection Agency (EPA), for example, has used for many years a VOSL range of $0.6 million to $13.5 million, with an average of $4.8 million (1990 US$) based on an assessment of the existing US literature (Robinson, 2007). However, whilst there is a very large body of research on health values for North America and Europe, there has been until very recently a dearth of evidence for developing countries.

This makes the gathering pace of current efforts to expand the VOSL dataset beyond developed countries all the more interesting. For example, a series of studies in China have focused on the potential aggregate benefits of reduced air pollution, primarily the avoidance of PM10 related deaths and associated diseases. Annual health damage costs equivalent to 6.5% of Beijing's GDP between 2000 and 2004 (Zhang *et al*. 2007) and damages of $29 billion in 2004 across 111 Chinese cities (Zhang *et al*. 2008) have been estimated. In addition, it has been estimated that potential corresponding health benefits from the implementation of low carbon energy scenarios could amount to $1.5 billion for Shanghai in 2010 (Chen *et al*. 2007). These studies contain no new VOSL data, instead relying on the transfer of the same 2001 Chinese CV study, which gave a figure of $44k, combined with internationally recognised dose-response functions. However, this value is at least in line with a recent Chinese hedonic wage study which estimated VOSL to be in the range $30,000 to $100,000 (Guo and Hammitt, 2009).

Other developing country hedonic wage studies have resulted in estimated VOSLs of $235,000 to $325,000 in Mexico (Hammitt and Ibarrarán, 2006), $375,000 in India (Madheswaran, 2007) and $790,000 to $2.41million in Poland (Giergiczny, 2008). A CV study based on reduced air pollution in Brazil produced a wide-ranging VOSL estimate of $770,000 to $6.1 million (Arigoni Ortiz *et al*. 2009), whilst a rural Thailand CV study produced a VOSL estimate of $250,000 (Gibson *et al*. 2007). Research on the impact of low levels of training in probability and risk concepts in Bangladesh finds significantly higher WTP values post-training (Mahmud, 2006).

This last point illustrates that while there is significant understanding of the use of valuation methods in this policy context, valuing health is still a complex proposition particularly when it implicitly or explicitly relies on people to value relatively small changes in risk. Concerns have focused on, for example, respondents' ability to comprehend small risks in stated preference studies (Corso *et al*. 2001, Hammitt and Graham, 1999), the possible existence of biases due to individuals heightened sensitivity to small changes in extreme health states (Bleichrodt and Eeckhoudt, 2006) as well as the nature of the relationship between monetary values and the scope (i.e. extent) of health gains from different reference points (Pinto-Prades *et al*. 2009): that is, for example, is the WTP of an individual for a reduction in some health risk from 1% to 0% the same as for a change in risk from 10% to 9%. Tests of preference imprecision (Vázquez *et al*. 2006, Svensson 2009), alternative theory frameworks to guide value estimation in practical studies (Abellan-Perpiñan *et al*. 2009) have also been the focus of research at the frontier.

Recent research, with practical implications for accounting, has shown that the age of the respondent who is valuing the risk matters (see Krupnick, 2007, for a review). While age may or may not be relevant in valuing risks which are immediate (e.g. life is threatened in the here and now), age has been found to affect the valuation of future (or latent) mortality risks. This is an important finding given that environmental contexts are associated with both immediate and future health risks. That is, the value of a reduction in the risk of death can be very different according to whether an environmental risk affects mostly older individuals already suffering from some underlying health condition (and who thus face a very present risk of dying as a result of exposure to say poor air quality) or younger individuals who might otherwise have enjoyed a normal life expectancy (in the absence of cumulative exposure to poor air quality). Thus, the standard practice of applying the same VOSL to value all reductions in mortality risk without regard to the age of those who benefit may be misleading. Strictly speaking, the damage caused by a general level of exposure to e.g. PM10 should be evaluated in terms of the (lower than immediate VOSL) valuations associated with younger people's valuations of future risks plus older persons' valuation of that risk as an immediate risk. An alternative approach that takes into account the age of persons saved by a particular policy, and that may be able to capture the shorter life expectancy phenomenon, is the value of a statistical life-year (VSLY) (see, for example, Hammitt, 2007).[12]

[12] The VSLY is calculated by dividing the value of a statistical life by discounted remaining life expectancy, thereby converting VOSL estimates into a value per life-year saved. VSLY can then be multiplied by life-years saved, i.e. the remaining life expectancy, to value the statistical lives of persons of different ages.

A rather distinct age-related issue is that some environmental risks fall disproportionately on the very young. One implication is that adults' (or parents') valuations of the risks on behalf of children need to be estimated with the finding in some studies so far of child/ adult WTP ratios of 2.8 in the UK and 1.6 in the Czech Republic (Bateman *et al*. 2009). Choice experiments carried out in Italy found tentative support for the notion that VOSLs were higher for children (Alberini *et al*. 2009). Efforts also have been made to yield valuations from actual behaviour by observing to what extent actions are taken to avoid potential (Mansfield *et al*. 2006) or mitigate existing health problems (Hanemann and Brandt, 2006).

There are alternative/ additional approaches to valuing health based on avertive or defensive expenditures as well as costs of illness (COI) and lost output. Common to each approach is the fact that they entail a financial outlay or implicit financial loss. Thus, in the case of avertive expenditure, goods might be bought which act as a substitute for a non-market good (e.g. air quality). However, individuals might change their behaviour in costly but perhaps less obvious ways in order to avoid an adverse impact on their wellbeing such as spending additional time indoors to avoid exposure to outdoor air pollution (Freeman, 2003). Nevertheless, the avoidance costs of spending time indoors could be evaluated by asking people directly about their time-use. Moreover, time use has a market analogue in the form of wages that would be paid to an individual if the time spent indoors could otherwise be spent working (see discussion of the travel cost method above).

Of course to implement this procedure we also require a value for the (shadow) price of time. One possible value is clearly the wage rate. If individuals can choose the number of hours they spend working then they will choose to work up to the point at which an extra hour spent at work is worth the same to them as an hour spent at leisure. At the margin, therefore, leisure time will be valued at the wage rate. In the real world, many individuals can only imperfectly choose the number of hours they work and the equality between the value of time in leisure and the wage rate is unlikely to hold. Empirical work has been undertaken that has revealed that time spent travelling is valued at somewhere between a third and a half of the wage rate and travel cost researchers frequently use one or other of these values as an estimate of the price of time (see, for example, Day, 2002).

The COI approach focuses on expenditure on medical services and products made in response to morbidity and other health effects of non-market impacts. For example, the costs of the health impacts of air pollution can be valued by looking at expenditure which affected individuals make on drugs to counter the resulting headaches, fever and other flu-like symptoms which some air pollutants are thought to cause. The difference between the COI and avertive expenditure approaches is that often the decision to incur these health care expenditures is not made by the individual alone, but by social administrators and ultimately the taxpayer. This can introduce ambiguities about what the COI approach is actually measuring for all its appealing simplicity. By contrast, when the focus is expenditure made by the individual, we can be (reasonably) confident that these expenditure decisions reflect the preferences of the individual for reduced negative impacts. However, expenditure decisions made by social administrators, politicians and so on might reflect other considerations including politics and ethics. A political decision to increase expenditure in a particular area might then appear to have made the problem worse (since costs of illness have increased), even though an individual's real health status might actually have improved (Pearce *et al*. 2006).

The difficulty with the COI approach can be that changes in expenditure on treatments of the health impacts of air pollution, for instance, are often not observed directly with ease. This can be the case for a number of reasons. Including the fact that the link between health and air pollution is stochastic, and that air pollution tends to cause health impacts which can arise for a range of other reasons. In these cases, the costs of illness are often calculated using an approach similar to that used for calculating lost output. The lost output approach is related to the COI/defensive expenditure approaches since it uses observed or estimated market prices as the measure of value. Examples in the case of health impacts include wages rates for changes in labour supply (because of a change in the prevalence of pollution related illnesses). The existence of this observed or estimated price is taken as evidence that a transaction (e.g. hire of labour time) would have occurred had the non-market impact (e.g. pollution) not had an effect. In fact, the negative pollution impact can be valued by estimating the resulting reduction in labour supply, and applying an appropriate price. Of course, one theme of all of these approaches relevant to the accounts is their shared emphasis on transactions. That is, either actual transactions which are currently recorded (in the case of COI/ defensive expenditures) or transactions which otherwise – and in the absence of a negative pollution impact – would have appeared in the accounts. The case for accounting for these values is based on making more explicit what these items ultimately are attributable to rather than measuring something that is wholly new to the accounts.

Progress in Valuing Ecosystem Services

Ecosystem services refer to the wide range of benefits that people derive from the multitude of resources and processes that are supplied by natural ecosystems (Daily, 1997). The Millennium Ecosystem Assessment (MA 2005), the most comprehensive survey to date of the state of the planet, indicates for example that nearly two thirds of the services provided by nature are in rapid decline. The next great challenge then for valuation practitioners is getting to a similar condition of understanding for ecosystems as arguably exists in the case of human health. This does not mean that little work has achieved in this respect to date.[13] On the contrary, a great many studies exist. In the case of forests, for example, advances have been made in measuring the economic values associated with timber and non-timber products, carbon sequestration and storage, recreation and watershed regulation. Progress has been more limited in estimating the non-use values of forests and a lively debate surrounds the value of genetic material in forests for pharmaceutical research (Pearce and Pearce 2001).

Hence, within particular ecological asset classes and given the multitude of services provided, the empirical record contains gaps. And not surprisingly, given the variety of ecological assets to being valued and the geographical breadth that needs to be covered, what is typically discovered by those who have sought to synthesise findings is that what exists is rather more like a patchwork of valuation studies rather than

[13] Turner *et al*. (2009) present evidence, over the last 10 to 15 years, of a substantial (in the region of at least one order of magnitude) increase in the number of published papers each year which use the term 'ecosystem services' (or similar).

anything approximating comprehensiveness. Moreover, ecosystem services are perhaps amongst the most complex environmental concepts to define, measure and value and there remain major methodological issues to be addressed (see, for example, Daily *et al*. 2000, Bateman *et al*. 2010). Significantly, many of these challenges are relevant to the domain of ecosystem accounting although few contributions make a clear link to the accounts in this way.

Haines-Young *et al*. (2009), however, provides the most explicit link to ecosystem *accounting* in trying to build a bridge between the discussions of ecosystem services in the wider literature and in the SEEA 2003 (UN, 2003). Specifically, UN (2003) makes reference to *resource functions* – or "… natural resources drawn into the economy to be converted into goods and services …" (p5) – *sink functions* – that "… absorb the unwanted by-products of production and consumption …" (p5) – and *service functions* – that "… provide the habitat for all living things …" (p5). What Haines-Young *et al*. seek to achieve is a cross-tabulation of a list of ecosystem services (reflecting the evolution of thinking about classifications since the publication of the MA) with this representation in UN (2003) as well as other official statistical classifications of economic activity, products and consumption expenditures. But, in large part, contemporary discussion (about ecosystem services) has been geared to scientific assessment and, increasingly, economic analysis (most notably cost-benefit appraisals and national ecosystem assessments) and the insights that a more explicit accounting approach could lend these efforts frequently have been lost. Clearly then this is a discussion upon which the accounting community can offer greater influence and guidance than is currently apparent.

That said, recent thinking about ecosystem services, in part, has drawn on recognition of the need for a consistency accounting basis for taking stock of diverse evidence. This thinking, in its turn, has centred on both the prices that might be needed to value ecosystem services but also and just as importantly has stressed the 'quantities' to which these prices are to be assigned (Boyd and Banzhof, 2007). Much of the debate about getting the physical accounting right inevitably has centred on the pioneering classification of ecosystems services in the *Millennium Ecosystem Assessment* (MA) (MA, 2005) the focus of which was: (a) *provisioning services* such as freshwater, food and so on; (b) *regulating services* including various types of regulation of climate, water, pests etc.; (c) *cultural services* spanning relatively tangible services such as recreation through to aesthetic and spiritual values; and, (d) *supporting services* such as primary production and nutrient cycling.

Since the advent of the MA, a number of contributions have sought to evolve this schema. This has resulted in something of a mini-battle over terminology as well as 'correct' classifications (although perhaps ironically many of these contributions call for consistency and protocols for consistency). Nevertheless, what is shared is a common objective to impose more structure on the ecological accounting problem. This has involved distinguishing between intermediate and final (ecosystem) services. Part of the motivation for this was the growing sense that the MA classification risked substantial double-counting. One recent classification in this vein, by Fisher and Turner (2008), is illustrated in Table 2. The distinction here is three-fold between not just whether services are (i) final or (ii) intermediate but also whether (iii) what is the ultimate benefit following a distinction first drawn, for example, Boyd and Banzhof (2007). This is more than just semantics. Circumscribing terminology in this way is means of ring-fencing the use of term 'benefits' for that thing that is closest to what enhances human well-being and is thus the thing that we want to value (e.g. food, drinking water, recreation, amenity etc.). Thus, as in the Table 2, food is a benefit which is derived from the final service of primary production (of ecological systems) which, in turn, depends critically on intermediate services such as pollination as well as abiotic inputs including sunlight.

Table 2: Classifying Ecosystem Services

Abiotic inputs	Intermediate services	Final services	Benefits
Sunlight, rainfall, nutrients	Primary productivity	Water regulation	Drinking water
	Pollination	Primary productivity	Food

Source: Fisher and Turner (2008)

It is perfectly possible, however, for a service to be final in one sense (primary production in the case of food production) and intermediate in another (primary production in the case of drinking water). A further complication is that some services might provide multiple benefits such as, for example, flood prevention as well as drinking water in the case of water regulation. Boyd and Krupnick (2009) call such instances 'dual commodities' while Turner *et al*. (2009) refer to this as 'joint production'.

Boyd and Krupnick (2009) have sought to make all of this thinking in the abstract about ecosystem services a little more schematic in terms of what they coin as an 'ecological production theory'. Again the crucial distinction is on ecological inputs and endpoints where it is the latter that ultimately gives rise to wellbeing by providing things that people might place a value upon receiving. What links these endpoints to inputs is some form of (ecological) production function. Critically, this production (and transformation of some combination inputs) might well be a complex phenomenon. However, the end-point itself may be more straightforward to understand: for example, it might be clean drinkable water. From the perspective of valuation, finding out exactly how beneficiaries value this end-point seems an uncomplicated task. Use of the term 'uncomplicated' here is relative and refers to the fact that people could be likely to be more familiar with the output than they are with the ecological production process (or the inputs) that yielded this end-point. Given that 'familiarity' is in no small part central to ensuring robust valuation then this is, in essence, a reassuring conclusion.[14]

[14] Nordhaus (2006), for example, cites an illustrative example from Nordhaus and Kokkelenberg (1999) of evaluating the marginal value of a loaf of bread – in the *absence* of a market price as guide – with recourse only to tracing the convoluted physical flows that go into the production of that loaf. The parallel to what these authors understandably describe as a "daunting task" is the measurement of the value of ecosystem services which currently lie outside of the market. The argument of many of those seeking currently to value the services provided by ecological assets is that just as it might be easier to find some way of asking people how much be willing to pay for the (unpriced) loaf of bread, so too could it be more straightforward to measure the final ecological output that is consumed.

Of course, we still might want to convey this end-value in terms of some more basic unit such as, for example, per area of ecosystem such as wetlands.[15] There might be a number of reasons for this. Conveying all of these values in the same (policy relevant) basic unit is a possibly useful means of allowing a coherent focus on the total value of a change in provision of a wetland. Importantly, wetland area – in this example – can be thought of as being the *asset* which, in turn, gives rise to services and ultimately benefits. More generally, Barbier (2009) outlines in detail a conceptual framework (and extensions) to thinking about ecosystems as assets and, in doing so, makes a case for using ecological land area as the basic unit of account (the unit in which physical services and the value of these services are cast).

The analytical problem is complicated by the need to know the link between the inputs and the output. That is, what is the precise relationship between the quantity or quality of inputs such as wetland area and the transformation of these inputs, via ecological production, into an output? Indeed, it may well be that the relationship between the extent of (or area covered by) the asset and the ecosystem services provided is not straightforwardly linear (see, for example, Barbier *et al*. 2008). The point stands, however, that it might be possible in a large number of cases (or certainly more cases than has been acknowledged previously) to separate this need for further understanding about ecological production from the valuation problem itself (which respectively can be left to experts and general public) (Boyd and Krupnick, 2009).

These issues of the getting physical and economic classifications right aside, the principles of valuation – for ecosystem services and ecological assets – are the same as for any other asset (and its services). However, as a practical matter, as Barbier (2007) puts it that these particular assets "… give rise to particular measurement problems [and this] is especially the case for the benefits derived from the regulatory and habitat functions of natural ecosystems." (p182). Following Just *et al*. (2004), Barbier goes on to note that ecosystems are unlikely to increase (and in sense are non-renewable), but rather these resources provide services which are largely unaffected by their use (and in this sense are renewable). However, what does threaten the future provision of these services is the ecosystem asset itself which can be destroyed by land conversion, habitat destruction and degraded through pollution. From the accounting perspective, this makes it worthwhile finding out what we enjoy from the ecosystems that we currently have and what we are losing when ecosystem assets are diminished.

Table 3: Land Use Values Per Hectare in Thailand, 1996-2004, US$

	(Net Present) Value per Hectare
Net income from collected forest products	$484 – $584
Habitat-fishery linkage	$708 – $987
Storm protection service	$8,966 – $10,821
Total	$10,158 – $12,392

Source: adapted from Barbier (2007)

[15] There is an analogy here between valuing the end-points of air pollution – such as increased mortality risks and illness – and the technical desire to understand and cast these values in terms of the damage caused by a unit or tonne of say particulate matter.

An example of what is lost is provided in Table 3 from Barbier (2007) for the case of land values for coastal mangroves in Thailand. These estimates, in turn, are based on viewing the value of this asset as being based on its ability to reduce the chances or probability and severity of economic damage. These values are interesting because they illustrate how an economic assessment would be affected by eschewing a fuller range of valuation methods. That is, those goods or services from mangrove assets which straightforwardly command market prices (forest products) apparently only constitute less than 13 per cent of the total asset value of a unit of this resource. The vast majority of this value then arises from the storm protection services provided by mangroves. The remaining component of asset value gives rise to the production of a further market (or near market) good. In this instance, however, the habitat-fishery linkage describes how a change in the habitat for commercially farmed fish not only affects harvest now but also – via dynamic effects impacts – the future productivity of the fishery.

Pascal *et al*. (2009) present a comprehensive overview of the state of valuation in ecosystem services. This review reveals at least three important points. The first is that a lot of effort already has gone into estimating the value of ecosystem services particularly elements such as recreation, carbon storage and so on. Nevertheless, a second point is that there are important gaps in this empirical record as discussed above for the case of the forests. Some of these gaps concern benefit categories based on, for example, non-use value although in this literature there has been some debate about how this notion fits the ecosystem service framework given that it is not purely a benefit flowing (either directly or indirectly) from an ecosystem asset or assets (Pascal *et al*. 2009). Rather, as argued by some, it is more of a social or individual construct and, indeed, may not emanate from experience at all (Fisher *et al*. 2009, Boyd and Banzhof, 2007). Less tangible components of 'cultural services', as defined in the MA, have been seen in this vein too. And given this evolution in thinking about classifying ecosystem systems it can also be asked how well do *existing* valuation studies of ecosystem services perform in terms of the compartmentalised thinking that these typologies recommend? Boyd and Krupnick (2009) argue that relatively few contributions appear to have anticipated these recent developments. Of course, what is just as interesting to ask is how this recent thinking can be used to guide the future conduct of studies. In this respect, there is emerging evidence in practical efforts such as the UK National Ecosystems Assessment that this is taking place (see, for example, Bateman *et al*. 2010).

The third point that emerges from the review in Pascal *et al.* is the extent to which basic frontier research issues tend to dominate concern about the evidence. Thus, for example, for all the progress made in classifying ecosystem services it is not altogether clear that in giving valuations (where relevant) people can unbundle in the way that is assumed. This is likely to be a particular problem where ecological assets give rise to multiple (final) outputs. More generally, concern about basing valuations on imprecise preferences has led to proposals for using deliberative valuation workshops or 'market stalls' (e.g. Macmillan *et al*. 2002; Alvarez-Farizo *et al*. 2007) where participants are given the opportunity to discuss the individual or collective value of the proposed change in a group context. Current evidence on deliberative and group approaches to valuation is limited, with only a small number of studies and rather small sample sizes (see, for example, Alvarez Farizo *et al*. 2007). Proponents hope, however, that these approaches might be used in applications involving poorly informed consumers as well as unfamiliar or complex changes such as biodiversity loss (Macmillan *et al*. 2002). Investigating such claims through wide-scale testing is a rich topic for further research.

Notably, however, many of these concerns present challenges are not only relevant for ecosystem valuation and relate significantly to the extent of basic understanding of the physical dimension of ecosystems and ecosystem change. That is, once we move beyond relatively simple indicators (such as ecosystem area, species abundance etc.) to the physical services provided by ecosystems these complexities present themselves in a more explicit way. Pascal *et al*. (2009), in this respect, distinguish between 'supply uncertainty' and 'preference uncertainty'.[16] It is the latter which is part of the valuation problem itself. The former is a more generic concern and is present regardless of the role assigned to valuation. We return to a number of these issues in the penultimate section of this paper. However, one immediate issue is the extent to which evolving research at the frontier be allowed to arrest development of 'simpler' elements of incorporating values within green national accounts that are considerably better understood.

4.4. Fit for Purpose and Beyond? Taking Stock of the Valuation Options

The past two decades has seen a proliferation of methods, and applications of those methods, that have sought to uncover, in a variety of ways, the value of environmental impacts (and non-market impacts more generally). The critical point to make here is that almost all developments have been accompanied by substantial critical reflection. Thus, the increasing use of these methods has resulted in, on the one hand, ever greater sophistication in application and, on the other hand, ever-present scrutiny regarding their validity and reliability. Any legitimate critique must therefore engage with this evolving thinking.[17]

A disproportionate amount of this critical assessment has accompanied the rise in popularity of CV. This assessment of the merits and limitations of this techniques (and its variants) and underlying conceptual framework has originated within the economics profession (e.g. Diamond and Hausman, 1994; Boyle and Bergstrom, 1999; Carson *et al*. 2001) but also beyond (e.g. Fischoff and Furby, 1988; Sagoff, 1994). Some of the

[16] One development arising out of this focus on uncertainty also emphasises the possibility of irreversibility – perhaps because funds committed cannot be 'uncommitted' or because other effects of the policy cannot be reversed – combined with the potential to learn by delaying some action. In environmental economics, this is mostly known as quasi-option value (QOV) although elsewhere, in financial economics, it has been called 'option value' or 'real options' (see, for example, Dixit and Pindyck 1994). One interpretation of the QOV approach is that it urges more caution about losing environmental assets such as ecosystems. Indeed, Pindyck (2007), for example, speculates that consideration of these values could be dramatic for the environmental context. Few empirical examples, however, exist to date and moreover the handful of studies that have sought to use the notion of QOV to explain conservation decisions appear to find that it could be empirically less important relative to other considerations such as accounting for the presence of global externalities provided by e.g. tropical forests (see, for example, Albers *et al*. 1996 and Bulte *et al*. 2002). The implications for accounting practice are further complicated by the fact that QOV is not itself a separate category of economic value (Freeman, 2003).

[17] There is a growing realisation that revealed and stated preference information is highly complementary and combining these data can enhances the unique respective strengths of these respective data whilst minimising their limitations (see, for example, Cameron, 1992, Adamowicz *et al*. 2002 and Kling, 1997).

critiques drawn from the latter are philosophical in nature and, as such, discuss concepts of 'value' and the appropriateness of relying on private economic value systems to inform public decisions (see Haddad and Howarth, 2006). Arguably, many aspects of these debates transcend the specific issue of the worth of CV and have broader ramifications if accepted. But the largest body of literature that has emerged from the CV debate has focussed on technical aspects, constructing rigorous tests of robustness across a variety of policy contexts and investigating and correcting for the presence of bias or theoretical anomalies (see, for example, Bateman and Willis, 1999). In fact, according to V. Kerry Smith, "Contingent valuation has prompted the most serious investigation of individual preferences that has ever been undertaken in economics" (Smith, 2006, p.46).[18]

Nevertheless, it is sometimes hard to escape a feeling that many critics are seeking to evaluate the worth of approaches such as CV relative to a criterion of perfection (Boyle, 2003). This is unrealistic not least because perfection does not characterise any empirical methodology or, indeed, actual market decision. Nevertheless, Sugden (2005) notes two opposing perspectives on this debate about 'fitness for purpose'. The first, and arguably the position that is most prominent within environmental economics, is that taken as a whole (and notwithstanding notable caveats) empirical findings so far largely support the validity and reliability of CV estimates – from well-executed studies – of the value of non-market goods. This emphasis on good studies is important and, in turn, has led increasingly to informal or perhaps even formal requirements for practitioners to follow, in some way, guidelines for best practice. The second perspective seeks to square the esteem in which valuation approaches increasingly appear to be held in many quarters with emerging evidence on anomalies about individual preferences (Hanley and Shogren, 2005). This is evident in the increasing transfer, and investigation, of insights from behavioural economics in research on environmental valuation. Of course, such behavioural anomalies might be characteristic of any (including market) behaviour not just behaviour in SP surveys. List (2005), however, presents evidence that preference anomalies are likely to be a matter of degree and, in part, are determined by experience and familiarity which is often lacking when environmental goods are not traded directly.

Clearly, these debates are relevant to deciding on the efficacy of non-market methods that ultimately might be used to green the national accounts. If a particular category of valuation method is unsuitable (or at least problematic) for the purpose for which it was designed then it is unlikely that it will be any more fitting for broader use. Unfortunately, the evidence does not allow any definitive judgement to be made and, if anything, might well suggest giving SP approaches the 'benefit of the doubt'. That is, the balance of the arguments suggests on the one hand a healthy dose of scepticism in the application, use and interpretation of any empirical methodology such as CV (Boyle and Bergstrom, 1999) and, on the other, the recognition that the elaborate methodology that has earned its place in the environmental economist's toolkit today is, in many ways, the culmination of constructive scepticism.

Nevertheless, the distinction between approaches based on market behaviour and those based on behaviour 'beyond the market' has been claimed to be at least suggestive of some broad hierarchy of valuation methods. These reflections typically

[18] By contrast, and with increasing exception, the CE format has received relatively little testing with respect to its vulnerability to empirical anomalies and biases in environmental applications (although, see Hanley *et al*. 2001).

begin with a reiteration of the fact that market prices are currently focal to valuation in e.g. the national accounts and so the further we move away from market-based approaches to valuation the more uncomfortable practitioners, in this context, might become. This view seems to be evident in the conclusions of, for example, the review of resource and environmental accounts in the US by Nordhaus and Kokkelenberg (1999) as well as, more recently, Stiglitz *et al*. (2009). Thinking about accounting for environmental values becomes, in part, an exercise in thinking about categories of things to be valued in terms of those goods and services which leave an indirect market trace that can be identified using one or other RP method and those whose value lies beyond the market either in a direct or indirect way and for which only SP approaches are available.

The crucial (and well-known) shortcoming of circumscribing valuation to these approaches is that "… revealed WTP is often an incomplete measure of a resource's total value" (Boyd and Krupnick, 2009, p38). One aspect of this incompleteness is the inability to estimate non-use values, as these methods are based on what is revealed largely by use-related behaviour. Another is the inability to estimate use values for levels of quality, or provision generally, that have not been experienced and then revealed by the market. With regards to non-use values, however, some commentators might argue that this is not a critical sacrifice of information in that, at first glance, their ethereal nature might justify leaving these out of the accounting problem. This could be justified on the basis, for example, of one criterion in Nordhaus (2006) that impacts to be considered in an accounting setting should be 'near-market' (although, having said this, 'non-use' would satisfy both Nordhaus' other criteria for inclusion which comprise policy relevance and evidence of a significant contribution to economic welfare). But even this judgement requires some further pause for thought. Clearly, some elements of 'non-use' are captured in the national accounts already through payments to conservation groups, heritage groups and so on. The benefits received by the work these groups do are imperfectly captured but clearly do not lie beyond the market altogether.

This emphasis on valuation methods based on market behaviour also raises the possibility of a paradox in that those methods which are judged to be most robust are those which are most likely to uncover values which are already in the accounts. On this view, the accounting issue is then to attribute these values to their true source (e.g. some environmental input or output). Without, however, conducting a detailed audit it is not really possible too much more about the extent of this issue. Nor have the assumptions that underpin RP been subject to anything like the scrutiny that has accompanied developments in say CV approaches (Bishop, 2003). The inherent assumption is that RP approaches are validated by the very fact that they rest on a behavioural component whereby a non-market good or service is implicitly traded. For SP it is evident that there is a difficulty of validation with reference to market data almost by definition. If a corresponding market-like price or market-based behaviour existed then the stated preference information would not be needed. Nordhaus and Kokkelenberg (1999) while preferring revealed preference data in greening the national accounts, argue that stated preference approaches should not be ruled out given the work being devoted to this area.

5. 'Scaling-up' Non-market Valuation to the National Accounts

5.1 The Inevitability (and Reliability) of Value Transfer?

While advances in methods that seek to generate primary data on the value of environmental goods and services has been a striking feature of modern benefit assessment, routine use of valuation in that context as well as in the national accounting domain arguably will rely just as heavily on using secondary data: i.e. the results of existing studies but applied to new (yet related) settings – so-called value or benefits transfer. This is a crucial issue as it is inconceivable that original valuation studies can be done for every service and every natural asset that would be needed for a comprehensive green national account. Even prioritising particular natural assets (perhaps on the basis of policy relevance and empirical significance) undoubtedly would represent too formidable a practical challenge. Value transfer thus represents a means of extending the empirical record in a pragmatic manner. Indeed, the holy grail of this approach is the consolidation of original data on non-market values in emerging transfer databases where values can be taken 'off the shelf' and applied to new policies and projects as needed (such as the web-based, Environmental Valuation Reference Inventory or, EVRI: see, www.evri.ca). The TEEB Review represents a more recent effort to assemble a database of (potentially) transferable values for a wide range of ecosystem services (www.teebweb.org). Invariably, however, transferring values in this way introduces an additional dimension of uncertainty into any valuation exercise in that it entails further assumptions and judgements to those contained in original studies.

The value transfer approach itself is the subject of a rapidly growing literature (see, for example, Boyle and Bergstrom, 1999, Desvousges *et al*. 1998, Navrud and Ready, 2007). In the policy appraisal context, what the approach involves is taking a unit value of a non-market good estimated in an original or primary study and using this estimate (perhaps after some adjustment) to value benefits that arise when a new policy is to be implemented. In the national accounting context, this suggests a further challenge. The 'new policy' – to which unit values from original studies might be transferred – is, for example, the total (i.e. national) change in some natural asset. We return to this issue of 'scaling-up' values below. Before doing so, it is worth briefly reviewing some elements of the recent discussion that has surrounded value transfer.

There is widespread recognition that the transfer approach raises considerable challenges, not least ensuring there is an abundance of good quality studies to populate e.g. valuation databases across the array of environmental changes that are of interest to decision-makers. Just as importantly, the validity of value transfer remains open to scrutiny. Indeed, a number of contributions have sought to test the accuracy of transfer exercises.[19] Reviews by Brouwer (2000) and Rosenberger and Loomis (2003) have sought to summarise the findings of a number of these types of test for recreational resources, water quality improvements and landscape amenities. Brouwer

[19] Ready *et al*. (2004), for example, undertook CV surveys in five European countries to elicit WTP to avoid health effects thought to be associated with air pollution. A test of the validity of benefits transfer was permitted by estimating WTP in any one 'policy' country on the basis of the values derived in the other (study site) countries. Comparison of the benefits transfer estimate with the actual value derived from the contingent valuation study in the 'policy' country provides a measure of validity of the transfer exercise judged on the basis of some (statistical or other) criterion or criteria.

and Bateman (2005) investigate the temporal reliability of transfer values: that is, the question they ask is at what point 'older' vintages of original studies are likely to become too unreliable to transfer to more contemporary environmental changes.

Distilling an overall message from these tests, however, is not straightforward. In some cases transfer error ranges are small while in other cases these ranges are extremely large indeed.[20] The evidence to date suggests that there is a need for still more research to secure a better understanding of when transfers work and when they do not as well as developing methods that might lead to transfer accuracy being improved. Intuitively, more sophisticated transfer approaches might be the answer. Such techniques seek to control for as many important differences as is possible between an original study site (or sites) and a new 'policy' site: e.g. relating to the attributes of the environmental goods as well as the socioeconomic characteristics of the populations at each site.

As for the case of original valuation studies, given the likely extent of some 'inaccuracy' the question is again one about can be said explicitly about the hurdle of precision if value transfer were to be used in extending valuation in efforts to green the national accounts. One problem that any transfer (and its conduct) faces is the lack of standardised approaches or ground-rules at the stage of conducting original valuation studies (despite emerging official 'guidelines' and 'manuals'). Even a relatively brief review of studies, in a limited number of environmental contexts, indicates the sheer diversity of the approaches that have been pitched at the valuation challenge.

This heterogeneity is clearly a marked contrast to the national accounting domain. Of course, this could be said simply to be an indicator of a (sub-) discipline that (relatively) is still in its formative years. However, Rosenberger and Johnson (2008) argue that some of this variety is attributable to the publication bias of journals where methodological novelty is at a premium while the more prosaic – but nonetheless important – matter of seeking to replicate past findings or extend the empirical record is at a considerable discount. Whatever the reason, this situation does hamper the task of making sense of the information contained in the mass of studies that currently make up the empirical record. Increasingly, structured and statistical approaches – known as meta-studies or meta-analyses – have been brought to bear on this challenge.

5.2 Meta-Analysis: Making Sense of the Empirical Record

Stanley (2001) defines meta-analysis as a statistical analysis where "… the dependent variable is a summary statistic, […], drawn from each study, while the independent variables may include characteristics of the method, design and data used in these studies" (pp131-2). There are a number of reasons for wanting to undertake meta-analyses of environmental valuation studies. One is to provide transferable values which we discuss in more detail below. Another reason is – as Stanley (2001) puts in

[20] In the case of health value transfers across national boundaries, for example, Barton and Mourato (XXXX) find large transfer errors for two comparable CV surveys eliciting WTP to avoid ill-health symptoms associated with exposure to polluted coastal water in Portugal and Costa Rica. By contrast, Alberini *et al*. (XXXX) find that transferring health values from the US to Taiwan provides a reasonable approximation of the findings of an original study conducted in Taiwan.

a general context – to reveal why there appears to be (perhaps wide) variation between the findings of original studies and, specifically, in a context relevant to the accounting problem to make sense of what values to use for this purpose.

The meta-approach is particularly relevant to environmental valuation where, perhaps not surprisingly, a wide range of distinct methods has given rise to wide range of estimates of values for the services provided by environmental and natural resources such as wetlands, forests and so on. If different studies are giving different valuation answers for the same (or broadly) similar category of good or service then this begs the question as to what is driving any divergence. In some cases, this might be that different valuation studies are seeking to measure distinct value concepts.[21] But it might also be that variants of the same methods give rise to divergent results because of differences in estimation procedures and study design. At first glance, this variation might seem bewildering and, at worse, another reason for distrusting the findings of such studies. At the very least this situation is unhelpful for advancing the use of values in informing policy thinking and, by the same token, encouraging the uptake of these findings in greening the national accounts.

The useful feature of meta-analysis here is that it offers a systematic way to understand study-to-study variation in valuation estimates for say wetland services where these differences exist. Indeed, a growing number of assessments have sought to do exactly this in respect of urban pollution, recreation, ecological services of wetlands, value of statistical life, noise and congestion (see, for example, Navrud and Ready, 2007). For the most part, these meta-studies use summary statistics, most notably willingness to pay values (Rosenberger and Loomis, 2003). Typically in individual studies, researcher seek to explain determinants of WTP with reference to characteristics of the environmental asset in the location being studied and the households or individuals that benefit from provision (Navrud and Ready, 2007). Meta-analyses, by pooling the summary findings of a number of studies, allow the characteristics of studies themselves to be examined as well.

This offers the potential of important new insights, perhaps most notably about the way in which differences in the methodologies used in original studies affect WTP values. That is, some of these studies may have relied on revealed preference techniques to uncover WTP values. Others may have used stated preference methods and within this category the potential for variety is similarly large with some studies using choice experiments and other using some variant of contingent valuation. Furthermore, these values might have been elicited in different ways using, for example, distinct payment vehicles and time frames over which payments are made, elicitation formats (i.e. the way in which the WTP question was asked) and survey modes (i.e. in-person interviews or some alternative). But the fact that studies uncovered or elicited WTP in different ways can be controlled for – in these meta-studies – and the influence on the variation in the resulting values better understood. Other factors which are likely to be important might include geography (where in the world the good or service being valued was located), the vintage of the study and so on (Navrud and Ready, 2007).

[21] This might be willingness to pay (WTP) or maintenance costs, WTP or willingness to accept (WTA), Hicksian or Marshallian demands and so on.

The strength of a meta-analysis is that it takes advantage of the collective wisdom embodied within a wide range of studies. And the emphasis on a statistical approach to taking stock of this accumulated knowledge adds a further useful dimension to 'simple' literature reviews of existing research (Stanley, 2001). Another interesting feature of meta-analytical approaches in this context is that it has focused attention once more on the quality of valuation studies. This issue is clearly relevant in the national accounting setting. That is, would the use of environmental values within an experimental account rely on new information gathered officially via statistical offices or would it rely (where evidence was available) on information already in the valuation literature? On the assumption that the latter is the more likely prospect (so as to avoid unnecessary burden and duplication), this raises the issue of what 'health checks' would need to be placed on valuation data drawn from available studies.

Within the meta-analysis literature, this debate has centred on those meta-studies which seek to generate values which can be used as a basis for value transfer (Smith and Pattanayak, 2002). One such issue is the extent to which any synthesis of the empirical record should restrict itself to the outputs of contributions appearing in peer-reviewed sources (e.g. primarily academic journals) or broaden the sweep to cover the findings in the 'grey' (or unpublished) literature. The instinct of most meta-analysts in the environmental valuation setting has been to cast the net (on what is legitimate evidence) relatively widely. This has the virtue of overcoming likely 'publication bias' whereby original studies which use current good practice but do not push the research frontier tend to be a premium in the peer-reviewed literature (see, for example, Rosenberger and Johnson, 2008). In fact many of the former have been produced by official agencies such as government departments (and will have been subject to their own validation and review procedures). A recent study by Nelson and Kennedy (2009) has sought to take stock of what might constitute best practice using this meta-approach to synthesising valuation evidence, and by this same token, to what extent has best practice been used in existing meta-studies of this type. One finding, perhaps not surprisingly, is that some of the problems with meta-studies (and so value transfer based on this research) can be traced back to the preliminary search for original valuation studies. The issue then of weaning poorly executed original studies from the sample to be analysed remains an important one.

Meta-Analysis and the Example of Wetland Valuation

Brander *et al*. (2008), in a study conducted for the European Environmental Agency, provides a review of meta-analysis from the perspective of its worth in coming up with transferable values for ecosystem services. Using the example of wetlands, their approach provides a good example of the strides that can be made in coming up with relatively sophisticated estimates of values per hectare (of wetland of different types). Just as interestingly, this study aims to scale-up these unit values so as to evaluate changes in wetland value at a relatively high level of aggregation (e.g. national or regional level).

The meta-analysis carried in this Brander *et al.* study itself makes use of a wide range of original research and identifies from a database of almost 400 valuation estimates obtained from a little less than 170 studies. However, only those studies which sought to value types of wetland that occur in Europe were included the analysis. Explanatory variables included study characteristics including valuation method, wetland type, size and service being valued as well as several other variables such as income per capita, population densities in the study countries. Values (e.g. WTP), the dependent variable in the meta-regression, is expressed as the value per hectare (in principle, describing what the population of the country in which the wetland is located would pay for a unit change in area of that wetland) and standardised in a number of ways (e.g. currency, purchasing power and prices). A range of these explanatory variables are found to be significant – in explaining the variation between values per hectare – including whether the wetland is a peatbog (which seem to command lower values, other things being equal), whether the wetland provides flood control and storm buffering services (which command higher values) and the size of wetland (larger wetlands appear to generate smaller – per hectare – values). Wetland values are also higher the richer the residents are, on average, of the country where an original study was conducted and the more people there are who live in the vicinity of wetlands in study countries.

Table 4: Proposed European Wetland Valuation Database

	Number of wetlands	Wetland area (ha)	Mean value/ per hectare/ year
Country			
Finland	14,140	1,971,961	224
France	1,419	358,163	5,693
Germany	1,391	418,945	4,353
Ireland	2,173	1,210,044	676
Netherlands	273	269,753	7,871
Romania	1,532	384,611	2,615
Sweden	20,242	2,729,131	263
United Kingdom	2,119	753,691	2,480
Wetland type (all Europe)			
Inland marshes	8,842	1,159,153	4,129
Peatbogs	38,644	6,712,309	214
Salt marshes	1,621	306,754	5,734
Intertidal mudflats	1,180	995,094	4,112
Salines	246	72,467	5,475

Source: adapted from Brander *et al*. (2008)

Understanding the sources of difference between values found in distinct studies in clearly interesting. However, a second objective of this work was subsequently to ‘scale-up’ and come up with defensible estimates that can be multiplied across many such areas of wetland across Europe. There are also important limits on this approach which we move onto discuss later. Given the relevance of the objective of ‘scaling up’ to the national accounting problem, it is worthwhile firstly setting out briefly the preliminary findings of this study. This includes a detailed breakdown of European wetlands by country and by wetland type. The basis for these values was the statistical analysis as previously described and information about (50,533) individual wetland sites across Europe (and e.g. their vicinity to populations etc.). This is presented in Table 4 which includes (in the upper part of the table) those countries with more than 250,000 hectares of wetland and (in the lower part of the table) a breakdown by wetland type (for Europe as a whole). So, for example, the findings indicate that peatbogs are the most abundant wetland type but command the lowest per hectare value. As such there is a resulting disparity in per hectare values between countries of wetland depending on the relative prevalence of wetland type in that country as well as, for example, demographic and socioeconomic factors.

To what extent are these estimated values reliable? The authors themselves outline a number of caveats. These include the fact that (as currently estimated), these values are relevant to evaluating changes which involve say a decrease in wetland area perhaps through land conversion rather than more intermediate changes in wetland quality. Similarly, any assessment such as this is reliant on the coverage of different components of value in the primary literature. Gaps in the original empirical record are then of course translated into 'incomplete' (meta-) value estimates. At any point in time, of course, there is little that can be done about a lack of comprehensiveness. The risk is, however, that these gaps become disguised (and forgotten about) in the presentation of summary unit values.

5.3 Future Challenges in 'Scaling Up'

How approaches that have sought to address 'micro' policy appraisal questions can be incorporated in 'macro' accounts is a critical issue for further discussion. In the case, for example, of ecosystems most of those proposals, and studies, which have sought to address this 'scaling up' issue have tended to generate unit (per hectare) values which appear to be designed for multiplying across potentially large areas. This raises a number of issues. For example, it is likely to be the case that – ecologically speaking – such areas could be in practice heterogeneous in terms of their relative contribution to ecosystem services (Barbier, 2007). In other words, geography matters in the sense that different locations across the land area covered by the ecosystem might provide different services of differing values. The issue then, in this case, is whether the values that we have do justice to this heterogeneity.[22]

A further issue is that simply adding to the area of an ecosystem is unlikely to have the same marginal value as an equivalent unit change to a smaller (but otherwise identical) ecosystem. The reasons for this different marginal value might be that it is the way that people view the valuation problem (i.e. more is significantly better when the reference point is a relatively small ecosystem area) or ecological productivity (Pascal *et al*. 2009). The tendency, however, is to assume that, as a practical matter, the value of ecosystem damage increases linearly. Put another way, what is presumed is that unit values are constant such that degrading twice as large an area of wetland is twice as bad in terms of the adverse outcomes that arise. It is fair to say that the valuation literature has only begun to scratch the surface of these critical issues. An exception is Barbier *et al*. (2008) that examines the non-linear relationship between mangrove area and coastal protection (from storm and similar damage). Specifically, these authors find that the *marginal* benefit that additional mangrove conveys – in terms of attenuating waves – declines the larger the initial mangrove area is.

[22] There are a number of reasons for this. Observed relationships, for example, based on distance-decay complicate matters in that some evidence exists for the proposition that WTP values are likely to decline the further away an individual (or a household) is from an ecosystem.

More generally, it is not clear that 'bottom up' approaches – whereby each type of service is valued separately and then the values are added or 'scaled up' to get some idea of the total economic value of the ecosystem – are capturing the 'whole' value of the ecosystem. Put another way, the value of the system as a whole may be more than the value of the sum of its parts perhaps because of complex ecological interactions (Arrow *et al*. 2000). Thus, Pascal *et al*. distinguish between what they term the infrastructure (or primary) value and the output (or secondary) value of ecosystems. The latter is arguably what has been the focus of discussions about the 'value of ecosystem services' in, for example, Fisher and Turner (2008) and Boyd and Krupnick (2009) and practical ecosystems assessments such as Bateman *et al*. (2010) and the TEEB review (www.teebweb.org). The former might include the value of the stock of ecological assets themselves and is related to the ability of ecosystems to absorb (external) shocks and stresses – i.e. its *resilience* – and still provide services (which, in turn, may be a function of the diversity of the ecosystem).

It is fair to say that practitioners have made considerably more progress in understanding secondary rather than primary values. However, basic conceptual issues are becoming better understood (Maler *et al*. 2009; Farley, 2008). Currently, for example, it appears that efforts to assess the secondary value of ecosystems can say very little, as a practical matter, about the importance of biodiversity. The focus on resilience, however, may be one means in which the significance (or otherwise) of diversity can be explored further (see, for example, Perrings, 2006). Hence, recent contributions have likened ecosystem management to concepts of asset portfolio management which until recently enjoyed prominence in financial economics (see, for example, Heal, 2007). That is, having a more diverse portfolio is of greater value for maintaining resilience in the face of the risks that different possible states of the world might present.

One interesting development that has emerged from theoretical work treats resilience as a stock (Mäler *et al*. 2007, Mäler, 2008). In other words, the ability of an ecosystem to withstand shocks has a distinct asset value which can be degraded (or enhanced) over time. Nonetheless, as things stand, practical valuation is far from any sort of comprehensive accounting for resilience. Walker *et al*. (forthcoming) is an exception and provides an example for agriculture in South East Australia where cultivation has led, via changes in the water table, to soil salinity problems. Valuation here, unfortunately, is a relatively complex business and extending this approach beyond largely illustrative examples is in its infancy at best. Indeed, Walker *et al*. are themselves extremely circumspect about using their empirical example in the 'real world' owing largely to apparent uncertainties about the scientific and economic data. Nevertheless, this approach breaks new empirical ground in the understanding the value of changes in ecological assets.

The impression then is that there is a considerable challenge in translating existing values (suited as these are to valuing certain elements of relatively small changes in ecosystem provision) to e.g. the types of larger-scale change that are argued to characterise current threats to ecosystems.[23] This view is proposed by Pascal *et al*. (2009) who argue that valuation estimates are currently able to say relatively little about how to value large-scale losses. A related issue is the fear that as ecosystem

[23] There is also increasing recognition of the challenges raised in trying to value non-marginal problems such as climate change (see, for example, Dietz and Hepburn, 2010).

degradation and destruction continues, the greater is the risk that we are seeking to evaluate changes which bring us closer to genuinely critical thresholds (see, for example, Walker and Meyers, 2004, on the different forms that such thresholds might take). Clearly, these are critically important concerns. At present, however, there is little of practical merit that can be said about this issue (although see Longo *et al*. 2007 for a study of the value that recreational users place on the approach of thresholds determining the extent of algal blooms in coastal zones of the north coast of Belgium). One issue, of course, is that we need some assessment of how far we might be from thresholds and so to assess the extent to which 'traditional valuation' certainly does have something useful to offer.

A more general issue is that the implication of valuing changes in natural assets is the need to say something about the value of the *future* services. An excellent review of the challenges that arise from valuing future costs and benefits is provided by Horowitz (2002). Clearly, we cannot possibly know exactly what future preferences will be – that is, what future people will value – beyond those things we feel confident will continue to be required for survival or basic functioning. The usual and pragmatic response to this uncertainty is to assume that future people have the same preferences as those living in the present. This means that, other things being equal, estimates of e.g. willingness to pay elicited or revealed in the here and now (and reflecting how current people value some environmental change) are used to value changes in the future. As an illustration, this might entail applying "today's" value of a statistical life as the shadow price of a fatality prevented in say 100 years' time because of climate change mitigation now.

Of course, there will be differences between – as in this previous example – marginal values now and marginal values in the future. One source of difference will be the discount rate used. Discounting involves attaching a lower weight to a given unit (say $1) of future benefit (or cost) than to an equivalent present unit.[24] The weights are determined by time itself and by a 'discount rate' which is expressed as a percentage. Discounting is justified by the assumption that it is what people do, because they are impatient and the second is the fact that capital is productive (i.e. can be invested now for some future return). The debate that ensued since the Stern Review illustrates the controversy that might surround the choice between any of the components of the *social* discount rate. A recent review by Gowdy *et al*. (2009) in the context of ecosystems and biodiversity illustrate that the issues there are likely to be no less controversial. While much of this discussion is frequently abstruse it has, of course, very real and practical implications for valuing natural assets and changes in the value of those assets. Recent contributions provide support for an increasingly influential view that the assumption of a constant discount rate in the conventional approach should be replaced, in some form, by time-declining discount rates (see, for example, Weitzman, 1998, Gollier, 2002, Chichilnisky, 1996, Frederick *et al*. 2002).[25]

[24] Discounting implies that the weight, w_t, to be attached to a gain or loss in any future year, t, is less than 1. More specifically, the discounting formula is: $w_t = \frac{1}{(1+s)^t}$ where s is the (social) discount rate. This discount factor, w_t, there shrinks as t gets larger (i.e. as gains and losses become more distant).

[25] The practical effect of this then is slow down the increase in the discount factor across time (relative to the case where the discount rate is constant).

The valuation problem raises distinct issues over and above how future values should be *discounted*. Horowitz (2002) notes that, even if the preferences held by present and future people are identical, there are at least two important determinants of values that might change over time. The first of these is *income*. If, for example, it is reckoned that future people will be richer than people now then marginal values should be adjusted to reflect the fact that future people will value the same change more highly. The basis for this would be an assumption that what is being valued is at least a normal good. A second factor is *environmental quality*. In this instance, if it is thought that environmental amenities will become more scarce in the future then it is plausible that the (marginal) value that will be placed on future losses of this amenity will be higher (than now).[26] To use the jargon, the impact of changing incomes and environmental quality means that, for example, changes in natural assets have distinct *effective* discount factors, an insight that goes back at least to contributions by, for example, Krutilla and Fisher (1974).

This issue about effective discount factors and, in particular, how marginal values change as a result of changing environmental quality itself segues into an ever present concern about practical estimates of the value of changes in natural assets is the extent to which values truly reflect the relative importance of different assets. There are a number of aspects to this but arguably the most prominent surrounds substitutablility between natural assets and other forms of wealth.[27] But while this debate is clearly critical, practical insight (rather than speculation) is itself rather scarce. One immediate problem in answering such a question, however, is that it typically involves working with a literature – and body of knowledge therein – which is at least one step removed from being straightforwardly operational.

Gerlagh and van der Zwaan (2002), for example, look at the conceptual case where substitution possibilities might be a function of the resource stock itself. That is, when the resource is relatively abundant, losses in that asset do not matter (as long as produced assets are built up in its place). However, after some threshold, substitution possibilities diminish rapidly. The implications for valuation are that the nature of the value of gains and losses in the asset are somewhat different depending on what side of the threshold we are on. The practical import of such theoretical insights has been demonstrated more recently by Hoel and Sterner (2007) and Sterner and Persson (2008). These papers argue, in a similar vein to earlier contributions such as Fisher and Krutilla (1974), that if a natural asset becomes increasingly scarce over time then its price, relative to other things that we value, will increase, perhaps dramatically.

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[26] Horowitz (2002) also reviews a number of psychological factors that plausibly could come into play when thinking about how future people might value, in particular, environmental losses. Among these is the notion of *reference dependence*. This refers to the observation that people have an inclination towards valuing changes relative to some reference point: typically, the current situation that they face. Of course, if environmental quality is itself changing over time then so is the reference point against which change is evaluated.

[27] This concern is typically characterized in terms of whether development should be weakly sustainable or strongly sustainable. While there is some debate about when exactly this terminology entered the literature, the main ideas can be found in Pearce *et al*. (1989). For *weak sustainability*, there is no special place for the environment as such. Put another way, it is the 'overall' portfolio of wealth bequeathed to the future that matters. As long as the real value of this portfolio is held constant it matters little that its constituent parts change over the development path. *Strong sustainability*, by contrast, requires that the environment is accorded explicit and special protection. There are a number of variants on this position. Most generally, it requires that 'natural wealth' should (in some way) be preserved intact through specific conservation rules.

Hoel and Sterner (2007) show formally how this intuition is confirmed in a model where limits to substitutability are examined explicitly. In doing so, they demonstrate that not only will scarcity result in an increase in the relative price of a natural asset but that the magnitude of this increase will depend, among other things, on the substitution possibilities between the natural asset and other forms of wealth.

6. Towards Practical Implementation of Valuation in the National Accounts

Moving toward incorporating valuation of environmental aspects in the accounts involves a number of challenges. It is likely that certain categories (of adverse impact or services) that plausibly we might want to account for are more ready for implementation than others. From a practical standpoint, it makes sense to begin with those elements for which the challenges are largely resolved. In no small part, these challenges will consist of whether the basic physical data are available, the link between the damage or service and its impact on human wellbeing is relatively well understood and for which valuation of the relevant end-points (e.g. benefits or costs) are widely available. In this respect, Table 5 outlines an initial hierarchy of the readiness – in terms of implementation in (experimental) national accounts – of a number of categories of either environmental degradation or services provided by elements of the natural environment such as ecosystems.

Before proceeding to a discussion of specific entries, a number of comments are worth making. First, the hierarchy presented is intended as an interim assessment only and so should be considered in that light. In addition, the identification of aspects of damage or services is arguably not ideal and some there is likely to be some overlap between categories (e.g. climate change damage and carbon sequestration – foregone damage – in ecosystems). Secondly, the elaboration of categories themselves is not exhaustive and, clearly, for example in the case of ecosystems there are different biomes that provide different and sometimes distinct services. The table illustrates crudely the case of forests although within this category it does not make sense to speak of forests where there are different forest biomes (e.g. tropical, temperate and so on). However, what is indicated in the table is offered as a way to describe the broad issues about the readiness of ecosystem valuation for incorporation in the accounts.

In terms of this 'readiness', the categories of damage or service are themselves divided into three groups or columns within the table.

The first group consists of those items for which it can be stated with reasonable confidence that not only can valuation can be readily implemented but also it can be relatively widely applied. What this means is that current knowledge and experience with valuation is substantial and, in addition, this experience has been undertaken in larger-scale settings (than small micro studies) and across a relatively large range of countries. It also means that the requisite physical data are largely in place as well.

Table 5: Preliminary Assessment of Readiness of Implementing Valuation in the Accounts

Damage/ Service	Valuation can be readily and widely implemented given current knowledge and data availability	Valuation is feasible/ used but not ready for wider implementation. Policy context likely to need in next 5-10 years	Valuation largely experimental and widespread use is still a long way off
Air pollution (Health)			
SO_2	X		
NO_X	X		
PM10	X		
PM2.5		X	
O_3 (ground-level)	X		
VOCs		X	
NH_3	.	X	
CO		X	
Lead		X	
Noise			
Annoyance		X	
Health			X
Water pollution			
Health		X	
Other impacts		X	
Climate change			
CO_2		X	
Other GHGs		X	
Land			
Soil erosion (on-site)		X	
Soil erosion (off-site)			X
Ecosystems (e.g. Forests)			
Provisioning: e.g.			
Food/ raw materials		X	
Water flow regulation			X
Medicinal			X
Carbon sequestration		X	
Erosion control			X
Habitat/ Supporting: e.g.			
Species/ biodiversity/ gene protection/ support			X
Nutrient cycling			X
Cultural: e.g.			
Recreation		X	
Aesthetic			X
Culture/ spiritual			X

Notes: SO_2 (Sulphur dioxide); NO_X (Nitrogen oxides); O_3 (Ozone); PM (Particulate matter, PM2.5 particles less than 2.5 microns in diameter; PM10, particles less than 10 microns in diameter); VOCs (Volatile organic compounds); NH_3 (Ammonia); CO (Carbon monoxide); CO_2 (Carbon dioxide); GHGs (Greenhouse gases)

The second grouping refers to those valuation categories which are not quite ready for wider implementation but where there is an institutional or policy context for wider scale valuation to be undertaken within a decade or so. For example, the likely increasing need to quantify in economic terms the impacts of European Union directives should lead to a corresponding increase in experience of using valuation in these domains more widely. It may be in such circumstances, that possible scope for incorporation is not that far away and so further investigation of feasibility is warranted in future work.

The final grouping describes those elements of damage or service for valuation is in all likelihood even further away in terms of readiness for implementation. In these cases, it is likely all that can be done is to maintain a watching brief on progress that emerges in future applications of valuation to these domains.

With regards to the specific (interim and tentative) demarcation of where items of damage or service fall in terms of 'readiness', it will be evident that a rather conservative approach has been taken. That is, only a handful of these items are indicated as being ready for widespread implementation. These correspond to what Hunt and Ferguson (2010) among others describe as the 'classical' air pollutants. The narrow range of items included in this grouping is in large part due to its demanding nature. In other words, all of the elements that require robust valuation must be in place currently. These range from good understanding and quantification of the physical pathway that results in exposure to either some environmental harm or service impacting on human wellbeing as well as their being a significant evidence base for that impact (or end-point) being valued. Not only this but this valuation must be capable of being 'scaled up' to the national accounting level as well as applied across as wide a range of countries as possible.

It is important, therefore, not to interpret the second grouping (column 3 in Table 5) as meaning that the categories of damage or service therein are not at all ready currently under any criteria. Put another way, this particular demarcation in Table 5 arguably does not do precise justice to where readiness (for these components of damages or services) is apparently lacking.

In some cases, it might be the physical evidence base that needs development in that valuation is relatively well understood. This might be the case for certain impacts arising from exposure to air pollutants where the value of the health effect or endpoint itself is relatively well established. Examples here could include fine particles (PM2.5) where the health effects associated with exposure are now comparatively well understood (see, for example, WHO, 2006). However, what might be lacking is the monitoring of that particular categorisation of particulate matter. This is likely to be a question of degree in that some countries are likely to have well-advanced monitoring (of e.g. concentrations) in this respect. Indeed, this is one area where advancements in knowledge and data can be relatively rapid (and the list of air pollutants which now fall within the domain of *routine* economic analysis is increasing all of the time).

Valuation is also well-understood in the case of some ecosystem services notably for provisioning services such as food and raw materials as well as an aspect of cultural services, namely recreation. In the case of the former, these goods are close to market and often may have then a commercial parallel on which to base valuation. In the case of the latter, valuation experience is relatively well-established (either using revealed or stated preference studies) and the service itself is something which is familiar to users and so valuations can be hoped to be correspondingly robust. The issue might be – respectively – the wider availability of physical data or the ability to scale-up or transfer the existing evidence base to all areas providing recreational opportunities. In some cases, however, the task that needs to undertaken in order to develop this evidence base may not be that be that great.

In the case of carbon dioxide and other greenhouse gases (GHGs), there are national inventories of physical emissions exist as well as increasingly good information in many countries about (net) carbon sequestration and storage in certain types of biomass. Likewise, estimates of the social cost of carbon (SCC) have been available for many years (see, for example, the meta-study of Tol, 2005). Moreover, these values typically are expressed as damage per tonne of GHG and, on the face of it, 'easily' can be applied irrespective of location of emission (or sequestration) source. The issue here is, of course, the well-known uncertainties about the value that the SCC could take. Thus, the possible ranges (for these per tonne values) are large and are crucially dependent on assumptions about the future path of global GHG emissions (e.g. the SCC is higher if a 'business as usual' path is assumed rather than 'strong, early action' by all nations). A number of countries are moving towards establishing (or have established) official values for the SCC. For example, the UK approach is based on an explicit judgement that this value should be consistent with the government's target for global GHG concentrations in the atmosphere. Thus, one element of this is to establish an official 'shadow price' based on an aspiration for a global target for concentrations (e.g. 450-550ppm) and then to estimate the marginal abatement costs for the UK associated with contributing to reaching that target. DECC (2009) indicates that this value is £52 a tonne of carbon dioxide equivalent in 2010.[28]

A large number of valuation studies have looked at the issue of water valuation from the perspective of the impact on recreation, aesthetic values, health status and non-use. Thus, the empirical record is large but it is also diverse while at the same time likely to be clustered around particular geographical contexts (e.g. studies mostly undertaken a handful of countries). For non-health categories of impact, the scope for value transfer across geographical boundaries may be small (that is, if the original context where values were estimated and the context where we would like to transfer are too different). In the case of health impacts of exposure to contaminated and polluted water, valuation of a number of crucial endpoints (notably gastro-intestinal illnesses) do exist (Hunt and Ferguson, 2010). Where data are also available on the link between exposure (of affected populations) to polluted water and number of incidences of these illnesses are available and reliable (and physical data on contamination available), such health values for water-related morbidity can be used in a similar way to the air pollution case.

The water context is also one where policy developments could well advance wider applications of valuation given the existence for example of the European Union Water Framework Directive (WFD). This legislation requires all water bodies achieve 'good ecological status' (by 2015) and has been accompanied by measurement effort directed towards extending inventories to assess number of water bodies of particular types (within

[28] In fact, the UK approach is a little more complicated than this in that it distinguished between non-traded and traded carbon. The latter refers to that carbon which is transacted via the European Emissions Trading Scheme. Valuation here is based around what is judged to be the 'likely' trading price and in 2010 this was reckoned to be £21/tCO_2e. In addition, low and high values for non-traded and also given (and these are respectively £26/ £78 and £12/£27). Shadow prices for e.g. emissions from 2030 onwards to assumed to be converge on the assumption that all sources of carbon are subject to trading.

water management catchment areas) on basis of whether have ‘good’, ‘moderate’, ‘poor’ or ‘bad’ status on ecological criteria (and chemical quality criteria). In some cases, this status is distinguished further by e.g.: diatoms, fish, pH, dissolved oxygen, phosphate etc. Fewer studies, however, have used very sophisticated approach to assessing how changes in these different water quality attributes affect valuations (although see Egan *et al*. 2008).

A less fine-grained approach for a UK study of the WFD looked at household willingness to pay (WTP) to move to ‘good ecological status’ using a stated preference survey (Baker *et al*. 2009). This estimated WTP to improve local water bodies (those in the respondent’s regional catchment) as well as national water bodies (likely to be mostly ‘non-use’). Practical issues include translating these values into what is usable in individual water catchment management areas (there are 11 of these in England & Wales) and so mapping available values onto available physical end-points.

Another development again in the European context, is the establishment of the evidence base, arising from the requirements of EU’s Environmental Noise Directive, of data on noise arising from road, rail, air and industry. For example, in the UK, such maps have been (are being) developed according to criteria to prioritise likely major sources of noise in 23 agglomerations around England. This noise is mapped within bounds (0-54.9dB; 55-59.9dB; 60-64.9dB; 65-69.9dB; 70-74.9dB; 75+dB). Exposure to noise of course also needs to be combined with maps of e.g. population densities and so on to assess numbers of people affected by noise within these bands. In principle, then these estimates of exposed populations can be valued using available studies such as those reviewed for the European context by Navrud (2004).

UK transport appraisal guidelines set down official marginal values for 1dB changes in noise from different levels: e.g. Household WTP to reduce noise from 46dB to 45dB, 61dB to 60dB and 82dB to 81dB is set at £8.4, £48 and £98 respectively. These values themselves are derived from a study of the relationship between rail and road related noise and property prices by Day *et al*. (2007). This is an illustration therefore where one valuation study was used as the starting point for scaling up to the national level (see, Nellthorp *et al*. 2005 for details on how this transfer was achieved). These values correspond to the annoyance that households endure as a result of exposure to noise levels in say cities. Less is known about the implications of exposure to high levels of noise on health although as Hunt and Ferguson (2010) note there is likely to be some (implicit) degree to which values based on ‘annoyance’ capture some of this health aspect.

In summary, a number of these categories of damage or service (in column 3) may be closer to being amenable to wider valuation than others. And, indeed, it could well be that for some countries (physical and monetary) data is available so as to make a particular category of impact ‘ready’ for incorporation in the national accounts in that location. Such instances would also make useful illustrative experimental cases to broaden applications of valuation in experimental accounts beyond a relatively small number of air pollutants. With regards to the final column in Table 5, these categories of damage or service are those where our preliminary assessment might suggest are relatively far from being available for widespread use. Typically, these involve elements where either few studies so far exist on which to base such rolling out, the valuation issues are complex or both problems exist.

Table 6: Prevalence of Values for Ecosystems Services (Based on: Pascal *et al*. 2010; Kontoleon and Mullan 2008)

Broad category/ Specific service		Ecosystem: Forests			Wetlands
		Number of studies	*Methods used*	*Geographical coverage*	*Number of studies*
Provisioning	Raw materials	28	Mostly market price or cost-based		7
	Food	26	Mostly RP, PF and market price	Mostly South America, some Asia	24
	Climate regulation	13	Mostly cost based		2
	Water	9	Mostly PF and cost-based		6
	Moderation extreme events	8	Mostly cost-based		6
	Medicine	6			1
	Soil formation	5			4
	Water purification	4			8
	Erosion prevention	4			3
	Air quality regulation	4			…
	Biological control	3			…
	Water flow regulation	2			5
	Genetic	1			…
	Pollination	1			…
Habitat/ Supporting	Gene pool/ species protection	10	Mix of SP and RP		3
	Biodiversity/ species support	4			2
	Nutrient cycling	…			2
Cultural	Recreation	59	Mix of SP and RP	Mostly North America & Europe	26
	Aesthetic	10	As above	As above	5
	Cultural	…			3
	Spiritual	8			1

Notes: Count of studies based on Pascal *et al*. (2010) for studies undertaken since 1990; Indications of geographical location of forest studies based on Kontoleon and Mullan (2008). These will not correspond directly to the more recent review in Pascal *et al*.

Not surprisingly, many of these items in the final column are ecosystem services for which as we have reviewed briefly previously in this paper the empirical record is currently evolving from a relativelylow base. Table 6 illustrates some of the issues in a little more detail for an expanded number of ecosystem services. Using two of the reviews that have contributed to an on-going international assessment (i.e. TEEB), (while not exhaustive) the table gives a rough assessment of the number of valuation studies conducted for different ecosystem services (for forests and wetlands) in the past twenty or so years. For some of these services, a relatively large number of studies exist: e.g. food, raw materials and recreation. For many other categories, very few studies exist at all. Moreover, for some of these categories (such as climate regulation), studies exist but the resulting values are based on cost-based methodologies (e.g. replacement costs) rather than conveying a signal about the value of the benefit provided by these services. The geographical distribution (as illustrated for the case of forest ecosystems) of existing studies is also an issue with some areas of the world having a paucity of studies.

This having been said, the preliminary designation (in Table 5) of widespread valuation given some way off carries with it an important caveat. The significant attention currently being directed towards adding to our understanding of the value of ecosystem services (particularly in the context of large-scale assessments such as TEEB), there exists scope for substantial advance in the near future. That is, what seems problematic and uncertain now may not always be so.

We end this section by returning to those items grouped under the ‘readily’ and ‘widely’ amenable to implementation column in Table 5. Table 7 contrasts the broad data availability issues for a selected few categories. A notable feature of the elements of air pollution damage is that scientific and economic analysis has been able to break down the impact pathway into manageable relationships which are relatively well understood.

The basic building blocks for these calculations are data on pollution concentrations at, for example, urban locations. In some countries, this concentration record is likely to be based on a mix of monitoring sites at various locations and mapping based on models of dispersion from emission sources. The reliability of these data will be a question of degree and notably will vary across pollutant. For example, Defra (2007) reports on the uncertainties arising from that most basic unit of information: emissions inventories. These range, in this UK case, from relatively small uncertainty for pollutants such as sulphur dioxide and nitrogen oxides (±3% and ±8% respectively) to moderate in the case of e.g. PM10, carbon monoxide and lead (-20 to +50%) and large in the case of certain types of hydrocarbons (-70 to +200%). Clearly, where data on concentrations relies on mapping the dispersal of pollutants from emission sources these uncertainties will translate into these data on pollutant incidence at locations. WHO (2005) reports that outside of North America and Europe, data on ambient concentrations of even common pollutants such as particulate matter tends to be sparse but that models (of concentration determinants at locations where data are available) can be used to infer the likely incidence of pollution for those locations where data are ‘currently’ lacking.

Table 7: Further Comments on Incorporating Valuation in the Accounts

	Physical data	Impact-pathway	Valuation	Comments
Environmental damages				
Example: Air pollution	– National inventories on emissions, concentrations – Augmented where data lacking by international organisations modelling of latter (e.g. World Bank, WHO etc.)	– Dose-response functions eg. World Health Organisation (such as WHO, 2005 and related sources) – Some national information may be available for certain countries – E.g. US EPA, UK Air Quality Management Strategy) but most will tend to be drawn from 'international' evidence	– National studies, official values (where adopted) used for appraisal – Cross-country reviews of valuation studies for mortality and morbidity – E.g. OECD on-going work summarised in Lindhjem *et al*. (2010)/ Hunt and Ferguson (2010)	– Where national studies of values not available, international transfer might be an option bit clearly not ideal – Such judgements about transfers implicit, however, in many applications of dose-response relationships – Most comprehensive data for all elements (physical/ impact-pathway values) for PM, O_3, $SO_{2,}$ and NO_2
Ecosystem services/ type				
Example: Recreation	– Number/ extent of natural recreation by type	– Data on numbers of visitors by recreational site	– Coverage good for some countries and makes use benefits transfer	– Suitability of valuation studies – E.g. more recent travel cost studies take account of impact of substitutes/ site quality on values, older studies do not)
Example: Wetlands	– National inventories, international inventories (e.g. for Europe compiled and managed by European Environment Agency) – Wetland type by area	– Uncertainty about precise relationship between e.g. wetland area and physical services provided	– Several meta-analysis of available estimates of value per unit area exist: e.g. Brander *et al*. (2009)	– Values are relatively highly aggregated: e.g. per hectare values summarise number of services which are not easily 'unbundled'

Air pollutants such as particulate matter have been linked with a number of adverse health outcomes. These can range from relatively mild impacts to more serious impacts such as breathing difficulties which require hospital admission and, in the limit, premature mortality. For an impact such as a hospital admission for respiratory illness, an individual might require complex medical treatments over a number of days, as well as a significant period of convalescence at home, restricting the individual's ability to work or otherwise continue their life normally. From the perspective of general health accounting, a dose (or exposure) response relationship describes the way in which exposure to given levels of pollution concentration of an affected population results in a given number of episodes of a health endpoint (e.g. premature mortality or hospital admissions because of respiratory illness).

In the case of the appraisal of a new proposal to tackle pollution, what we would be interested in is the change in pollution concentrations from some current level. For accounting purposes, however, what is arguably relevant is the damage that arises from the entire exposure to concentrations (although it might be appropriate to deduct from this any natural background levels of e.g. dust particles in measures of PM concentrations for which 'policy' can do little to influence).[29] The dose-response function itself is based, in turn, on scientific evidence about the relationship between human exposure to a pollutant and its adverse consequences for health outcomes. In the case of PM, for example, exposure has been shown to be associated with decreased life expectancy, lower respiratory symptoms and decreased lung function in both children and adults as well as COPD (chronic obstructive pulmonary disorder) in the latter (WHO, 2005). The fact that some types of health outcome affect some groups rather than all groups clearly has a bearing on the scope of the population at risk from that particular end-point.

Examples of dose-response functions exist for a reasonably large range of health end-points particularly for sulphur dioxide, nitrogen dioxide, (ground-level) ozone and particulate matter (either fine particles such as PM2.5 or broader measures such as PM10). The range of these functions is, however, constantly evolving as 'novel' pollutants come within the ambit of this impact-pathway measurement and the specific role of individual pollutants becomes better understood (that is, for example, what is the contribution of nitrogen oxides to health outcomes over and above their role as precursors to ozone formation and nitrate particles in PM). Needless to say, therefore, at any point in time there are uncertainties in these functions. Indeed, Hunt and Ferguson (2010) cite evidence that uncertainties inherent in dose-response functions could be on a par with the uncertainty reckoned to be associated with the monetary values that are commonly elicited in stated preference studies of health outcomes. This does not mean of course that this dose-response information is not valuable but it is worth acknowledging once more that valuation is not the only source of uncertainty in evaluating (as here) pollution impacts.

Valuing the physical end-points that estimated from utilising information about dose-response relationships in combination with an assessment of the population at risk from an adverse health outcome is the final piece in this accounting puzzle. A recent review by Hunt and Ferguson (2010) indicates that a large number of studies have been conducted into a correspondingly large number of morbidity-related health end-points (ranging from minor symptoms such as coughing through to chronic bronchitis in adults, hospital admissions,

[29] In addition, given that certain pollutants can travel large distances on wind currents, ambient concentrations may include pollutants from emission source which originate in other countries.

types of cancer and premature mortality). This indicates that some care must be taken about the potential for overlap between these health categories (and thereby double-counting). Moreover, the authors of that review indicate that a significant proportion of the likely costs of adverse health arising from exposure to air pollution might be captured by focusing on a handful of these end-points (particularly, premature mortality and hospital admissions for example). Interestingly, on Hunt and Ferguson's reckoning, these are also typically the health end-points where the quantity and quality of valuation information is relatively good.

7. Concluding Comments

We began this paper by making the claim in effect that 'from-the-outside-looking-in' (and not withstanding important contributions made in a great many other respects), on-going efforts to green the national accounts have been largely immune to the 'valuation revolution' elsewhere. Given that one area where this growing emphasis on non-market valuation is most evident – in the context of public understanding of aspects of life such as the environment and health – is in policy thinking, there is a real risk that what is being sacrificed to preserve immunity is relevance.

Of course, there is a corresponding risk of over-claim here as there are a number of reasons why we ought to be sceptical about approaches that seek to put a value on non-market goods and services. Many of these reasons, as we have reviewed, have been raised in the literature on greening the national accounts and beyond. These debates are important and, if anything, methods have become stronger as a result of such dialogue. That said, it is crucial that valid scepticism is distinguished from critiques that have a less firm basis (or at least represent an inconsistent perspective in the context of the accounts). Clearly, questions about 'accuracy' and 'imprecision' inevitably loom large. But such questions should not simply be posed in the abstract. Explicit guidelines as to what hurdle of accuracy is to be passed thus have to be set if this concern about 'difficulty' and 'uncertainty' (of non-market valuation methods) is to be lend itself to constructive insight.

However, even concluding that non-market valuation approaches are defensible on their own merits and that there are valid grounds – in principle – for including such values in experimental green national accounts (i.e. no insuperable inconsistency), significant challenges remain. Most prominently, non-market valuation methods have not evolved with the aim in mind that say environmental values uncovered or elicited are used in *national* accounts. These methods typically have sought to inform choices over policies or projects which involve smaller changes in the provision of some natural asset. Thus, one very real issue is to move the accounting debate onto whether and how to 'scale-up', for example, ecosystem values in a justifiable and robust way.

These particular challenges may not be insurmountable and, in the preliminary assessment provided here, it may be that a number of categories of damage or service plausibly could be considered ready for valuation in the accounts and an even greater number in arguable 'near' readiness. Nonetheless, it remains important of course that accounting challenges are acknowledged and distinguished from those concerns about the merits or otherwise of valuation 'in the round' in order for a plausible way ahead to be identified. This raises a number of issues which are genuinely at the valuation frontier, give pause for serious reflection, and (at least for some of these matters) it may be that all that can be done is to maintain a 'watching brief'. Nonetheless, just as progress has been witnessed in other domains to which valuation has been applied, here too advance should be expected and issues to evolve. At the same time, neither should our wait for evolving research at the frontier be allowed to arrest development of ‘simpler’ elements of useful work in incorporating non-market values – within green national accounts – that currently are considerably better understood.

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